Title

Summary of the Nonmonetary Exernalities of Motor-Vehicle Use: Report #9 in the series: The Annualized Social Cost of Motor-Vehicle Use in the United States, Based on 1990-1991 Data

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SUMMARY OF THE NONMONETARY EXTERNALITIES OF MOTOR-VEHICLE USE

Report #9 in the series: *The Annualized Social Cost of Motor-Vehicle Use in the United States, based on 1990-1991 Data*

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This report is one in a series that documents an analysis of the full social-cost of motor-vehicle use in the United States. The series is entitled *The Annualized Social Cost of Motor-Vehicle Use in the United States, based on 1990-1991 Data*. Support for the social-cost analysis was provided by Pew Charitable Trusts, the Federal Highway Administration (through Battelle Columbus Laboratory), the University of California Transportation Center, the University of California Energy Research Group (now the University of California Energy Institute), and the U. S. Congress Office of Technology Assessment.

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LIST OF ACRONYMS AND ABBREVIATIONS AND OTHER NAMES

The following are used throughout all the reports of the series, although not necessarily in this particular report

AER = *Annual Energy Review* (Energy Information Administration)
AHS = *American Housing Survey* (Bureau of the Census and others)
ARB = Air Resources Board
BLS = Bureau of Labor Statistics (U. S. Department of Labor)
BEA = Bureau of Economic Analysis (U. S. Department of Commerce)
BTS = Bureau of Transportation Statistics (U. S. Department of Transportation)
CARB = California Air Resources Board
CMB = chemical mass-balance [model]
CO = carbon monoxide
dB = decibel
DOE = Department of Energy
DOT = Department of Transportation
EIA = Energy Information Administration (U. S. Department of Energy)
EPA = United States Environmental Protection Agency
EMFAC = California's emission-factor model
FHWA = Federal Highway Administration (U. S. Department of Transportation)
FTA = Federal Transit Administration (U. S. Department of Transportation)
GNP = Gross National Product
GSA = General Services Administration
HC = hydrocarbon
HDDT = heavy-duty diesel truck
HDDV = heavy-duty diesel vehicle
HDGT = heavy-duty gasoline truck
HDGV = heavy-duty gasoline vehicle
HDT = heavy-duty truck
HDV = heavy-duty vehicle
HU = housing unit
IEA = International Energy Agency
IMPC = Institutional and Municipal Parking Congress
LDDT = light-duty diesel truck
LDDV = light-duty diesel vehicle
LDGT = light-duty gasoline truck
LDGV = light-duty gasoline vehicle
LDT = light-duty truck
LDV = light-duty vehicle
MC = marginal cost
MOBILE5 = EPA's mobile-source emission-factor model.
MSC = marginal social cost

MV = motor vehicle
NIPA = National Income Product Accounts
NO_x = nitrogen oxides
NPTS = Nationwide Personal Transportation Survey
OECD = Organization for Economic Cooperation and Development
O_3 = ozone
OTA = Office of Technology Assessment (U. S. Congress; now defunct)
PART5 = EPA's mobile-source particulate emission-factor model
PCE = Personal Consumption Expenditures (in the National Income Product Accounts)
PM = particulate matter
PM_{10} = particulate matter of 10 micrometers or less aerodynamic diameter
$PM_{2.5}$ = particulate matter of 2.5 micrometers or less aerodynamic diameter
PMT = person-miles of travel
RECS = Residential Energy Consumption Survey
SIC = standard industrial classification
SO_x = sulfur oxides
TIA = *Transportation in America*
TSP = total suspended particulate matter
TIUS = *Truck Inventory and Use Survey* (U. S. Bureau of the Census)
USDOE = U. S. Department of Energy
USDOL = U. S. Department of Labor
USDOT = U. S. Department of Transportation
VMT = vehicle-miles of travel
VOC = volatile organic compound
WTP = willingness-to-pay

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9. SUMMARY OF THE NONMONETARY EXTERNALITIES OF MOTOR-VEHICLE USE

9.1 EXTERNALITIES: DEFINITIONS AND PRESCRIPTIONS

The literature on externalities is enormous, and there have been extensive debates on terminology and particular aspects of externalities (e.g., Freeman, 1984; Bird, 1987; Peskin, 1988; Bird, 1988; Oates, 1988; Freeman, 1988). What follows here is a brief sampling of some of the historical and recent thought on externalities, and an overview of current thinking on how they should be addressed.

9.1.1 Definitions

As noted by Baumol and Oates (1988), and Bator (1958) long before them, definitions here are a matter of taste and convenience. There is no "right" definition; at best, there are widely accepted definitions.

Decades ago, Bator (1958) distinguished several kinds of externalities: "ownership" externalities, which are attributable to a lack of property rights, or ownership; "technical" externalities, which are due to increasing returns to scale, and include the problems of decreasing long-run marginal costs and natural monopolies; and "public-good" externalities. This definition includes virtually all forms of market failure, and is broader than most economists now accept. Before him Scitovsky (1954) discussed several concepts of "external economies," including a distinction between "technological external economies" and "pecuniary external economies" taken from a 1931 paper by Jacob Viner.

Later, Arrow (1969) offered a definition narrower than Bator's (1958). He contended that externalities are a particular kind of market failure, and that the problem of increasing returns to scale is not a problem of market failure. He then pointed out that market failures can be analyzed in terms of the more general notion of transaction costs: if transaction costs are too high, a market will not exist. At the same time, Davis and Kamien (1969) made the now-usual distinction between pecuniary externalities (those that result from price changes only) and technological externalities (those that affect actual production possibilities), but defined externalities, quite broadly, to be effects on persons that are not associated with specified purchases.

More recent writers have continued to refine the definition of an externality. Randall (1981) says that an externality is present when some of the benefits or costs of an action are external to a decision maker's calculus. He states further that if the affected party would like the acting party to modify the action, then the externality is "relevant," and that if the activity can be modified so as to make the affected party better off without making the acting party worse off, the externality is "Pareto-relevant". He appears to attribute externalities to the absence of relevant property rights, and distinguishes this problem of "non-exclusiveness" from other fundamental sources of market failure or inefficiency: indivisibility in consumption, which gives rise to public goods; congestible goods, which are indivisible up to a point, called their carrying capacity, but congestible beyond; and monopolies, including "natural monopolies" with declining, rather than rising, long-run marginal-cost curves.

In their widely used text on environmental economics, Baumol and Oates (1988) state that an externality is present when agent A chooses the value of [a] non-monetary variable[s] in agent B's utility or production relationships without particular attention to B's welfare. This definition is useful. Note that it comprises two conditions: A affects B, but does not recognize or account for the effects. Thus, by this definition, a negative "externality" is synonymous not with "damage," but with "unaccounted for cost"[1]. (Of course, there can be positive externalities, too.) Baumol and Oates (1988) also argue that their definition rules out cases in which somebody *deliberately* does something to affect B's welfare. Thus, externalities are *unintended* effects. This distinguishes external costs from the costs of crime, for example. Whereas the prescription for externalities might be an optimal tax, the prescription for crime is enforcement of the law and moral suasion.

It is important to understand that external effects actually change production or utility relations, and thereby have real resource costs or benefits. By contrast, price effects, or pecuniary effects, merely shift the market equilibrium to different points along production and utility frontiers, without changing the underlying production and utility relationships themselves.

The Baumol and Oates view of the meaning of "externality" seems to be accepted today by most economists. I adopt it here.

Externalities versus environmental costs. It is worth noting that environmental costs are not necessarily externalities. In fact, one can distinguish three classes of environmental costs, only one of which falls in the class of externalities.

1). First are those costs that are imposed by party A on party B but not accounted for by A. These are externalities, which may be either monetary (e.g., some kinds of mitigation costs) or non-monetary.

2). But party A could be made to account for her damaging activities, most efficiently via an optimal damage tax. Then the damage would be accounted for, and hence no longer would be an externality according to our definition; rather, it would be an efficiently priced cost to the user. For example, we could set a price on emissions of particulate matter, equal to the expected cost of the mortality and morbidity caused by PM emissions at particular times in particular places. In this case, we would classify the expected health costs as a priced cost of motor-vehicle use, and not as an external cost.

However, such correctly priced environmental costs still should be distinguished from normal private-market costs, such as the cost of car tires, because in the normal market the price is set on the basis of private production costs and private demand,

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[1]"Cost" here includes the cost of defensive behavior as well as the cost of unmitigated or residual damages. For example, if motor-vehicle air pollution forces some people to stay indoors and others to wear masks, we should count as an external cost of motor-vehicle pollution the cost of the masks and the inconvenience of the masks and the indoor confinement, as well as the cost of any health problems caused in spite of the defensive behavior.

whereas in the environmental market prices would be set by a public authority. Hence, costs covered by Pigouvian taxes should be a separate cost category in the social-cost analysis. Presently, though, there are no correctly set Pigovian taxes on environmental damages caused by motor-vehicle use, and so I have not bothered to classify them separately in Table 1-1[2].

3). Finally, there are damages, such as motor-vehicle noise, imposed by party A on herself. In this case, party A and party B are the same; hence, if party A is aware of and accounts for the cost that she imposes on herself, then the cost is not an externality. (I classify these as "personal nonmarket costs".)

9.1.2 Prescriptions: the proper remedies for externalities

Table 1-2 presents a hierarchy of efficient treatment for externalities, which can be summarized as follows: 1) if possible, assign true micro-level property rights; 2) otherwise, try collective bargaining; 3) otherwise, enact dynamic Pigovian taxes. I will elaborate on each of these.

Externalities are usually attributed to the absence of property rights for the resource in question. For example, air pollution is an externality because air is not owned and bought and sold in markets. It follows from this characterization that the ideal or first-best remedy, in principle, is to establish property rights. Of course, in most cases, this is practically impossible, which is why usually there are no property rights in the first place. Nevertheless, we gain insight into the causes and consequences of externalities when we imagine what the world would be like if meaningful property rights could exist. In the following paragraphs, I distinguish two levels of property-rights: micro-level or individual rights, which allow each individual to consume exactly as much pollution as he or she pleases, and collective rights, which result in an optimal quantity of pollution for the collective as a whole.

In a perfect world, there would be a market in air pollution, and it would function just like the market in, say, bananas. This perfect (and unattainable) market would not be publicly run or collectively negotiated; rather, it would be a market of voluntary transactions between individual buyers and sellers. To function properly, this would have to be a market in molecules. Each molecule of pollution would be owned,

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[2]There is, however, at least one incorrect (non-optimal) charge for environmental damage related to motor-vehicle use: the part of the oil-spill-liability excise tax on petroleum that is used to compensate for damages from oil spills. The charge is not optimal because it is not set to reflect marginal damages, but rather to raise revenue to compensate for *some* of the damages of oil spills. The gas-guzzler tax also perhaps can be viewed as a non-optimal charge for environmental damages of oil use. Technically, this tax and perhaps the gas-guzzler tax should be included in a separate category called "non-optimal taxes for environmental damages." (The category "optimal taxes for environmental damages," as well the category "efficient charges for government infrastructure and services," would be empty.) However, these taxes are so minor that it is not worth making the separate accounting. Instead, I have classified them with non-optimal taxes for government-provided infrastructure and services.

bought, and sold, just as bananas are[3]. A polluter would offer to pay an individual to dispose of a specified number and kind of molecules in her body; the individual would assess the risks to herself of being a human dump or incinerator, compare it to the prevailing price, and decide whether or not to accept. If the person agreed and accepted the molecules for disposal, she would store them indefinitely in her body tissues, or metabolize them to harmless products, such as water vapor. She could not re-emit the pollutant (as people actually do when they breathe) herself without buying a disposal option from another disposal source. Every molecule of pollution would be identifiable and traceable. Individuals would have molecular detectors that checked each molecule of air that they consumed; if any unauthorized pollutant molecules trespassed, the owner of that molecule would be subject to the usual legal sanctions.

This example is absurd, of course, but revealing nonetheless: it shows the extent of the deviation of the real world from an ideal economic world. In a truly ideal market, everyone would consume exactly as much pollution as he wished, when he wished. It is sobering to realize that the best imaginable *real* world -- one with efficient pollution taxes -- probably would bear little resemblance to that ideal world, and in fact undoubtedly would be far inferior to it[4].

If it is not possible to establish true, micro-level property rights, then collective bargaining is the next best solution. In a seminal paper in 1960, Coase (1960) argued that, if bargaining is costless (or nearly so), it will lead to efficient handling of externalities. Otherwise, he suggested, there is no single universal prescription that always will lead to an optimal outcome. Generally, bargaining is feasible only when there are relatively few perpetrators and victims -- the so-called "small-numbers case". Collective bargaining obviously is inferior to individual choice, because only with complete freedom of choice does the individual always get exactly what he wants when he wants it.

Two aspects or implications of Coase's work should be mentioned. First, Coase showed that, if the costs in question are small relative to incomes, and if transaction costs are small, then it does not (or in theory should not) matter which party in an externality negotiation has the right of action -- that is, it does not mater if the polluter has to pay to pollute, or if the victims have to pay to prevent pollution; the resulting amount of activity will be the same in either case[5].

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[3]Note what we have done here: we have shed the usual reasonable practical assumption that air is a public (nonrival) good, and pushed on resolutely to the molecular level, at which, technically, we no longer have a public good: the molecule that I breathe is not available for you to breathe.

[4]In the Pigovian world we might end up with the right pollution prices and the right total quantity of pollution, but undoubtedly not the optimal allocation of that total across individuals. In a Pigovian world where nobody can pick his or her optimal quantity, most will be forced to consume more or less than the optimum.

[5]Coase's theorem in effect is a presumption that if the transaction cost is zero and the cost of damage is very small relative to income, then willingness to pay (WTP) to avoid damage should about equal willingness to accept (WTA) compensation for bearing damage. Interestingly, however, contingent

Second, it is now accepted that Pigovian taxes should be asymmetrical: that the perpetrator should be made to account for the marginal cost of her action, but the affected party should *not* receive compensation from the perpetrator (Cropper and Oates, 1992; Baumol and Oates, 1988). Rather, the affected party should be left to seek proper defensive measures. As Coase (1960) showed, the greatest social welfare might well obtain when the potential victims mitigate the prospective damage or avoid it altogether[6].

Turvey (1963) summed up this early work by Coase (1960) and by others as follows: when negotiation is possible, public intervention is unnecessary; when it is not (usually because of large numbers), "the theorist should be silent and call in the applied economist" (p. 313).

In principle, however, if negotiation is not possible, then Pigovian taxes are called for. (The non-trivial problem in practice is to estimate the optimal tax!) Baumol and Oates (1988) give the definitive prescription:

> In sum, irrespective of whether the externality is of the depletable or undepletable variety, the proper corrective device is a Pigovian tax equal to marginal social damage levied on the generator of the externality with no supplementary incentives for victims (p. 23).

They note further that "the damages that victims suffer from the detrimental externality provide precisely the correct incentives to induce them to undertake the efficient levels of defensive activities" [p. 22].) We would add that, given their definition of externality (see above), Baumol and Oates really mean that the externality tax should

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valuation studies of environmental amenities have not always borne this out: in some studies, hypothetical WTA is much greater than WTP, even for amounts small relative to income (Cropper and Oates, 1992). Analysts have offered three explanations of this discrepancy. First, the individuals surveyed might perceive that "the private market goods available in their choice set are, collectively, a rather imperfect substitute for the public good under consideration" (Hanemann, 1991, p. 646). Second, individuals simply might be more familiar with buying than with selling (Cropper and Oates, 1992). Third, it actually might matter to an individual whether initially he owns the good in question. Put another way, individuals might be averse to losses per se (Hanemann, 1991; Cropper and Oates, 1992). This last has been called the "endowment effect".

[6]It is easy to think of an example. Suppose that a woman is considering taking a relatively unimportant trip, which she values at only $2 above her total time-plus-money cost of driving. She is planning the trip at an especially dangerous time to drive, for there is a 1-in-100 chance that someone will crash into her and cost her $500 in pain, suffering, and lost time. If she knows that she will be fully compensated by the perpetrator, she will take the trip regardless of the risk. But society would prefer that she not take the trip, because it has a negative social value: the expected accident cost of $5 ($500/100) is greater than her consumer surplus (over her non-accident costs) of $2. Only if she has to bear the expected accident cost herself, either directly or indirectly via defensive insurance (per trip), will she not take the trip, which is the socially superior choice. Thus, compensation can encourage economically inefficient behavior. (Lack of compensation per se will never lead to socially inefficient behavior.) Baumol and Oates (1988) show this formally.

be equal to the *unaccounted for* marginal social damage, where "unaccounted for" is meant as discussed above.

Note that the optimal taxed is to be assessed on the "generator of the externality" -- that is, on the immediate damaging activity, and not on some related activity. In the case of air pollution, the tax should be levied on the source of the emissions. For example, the environmental damages from pollution from petroleum refineries should be internalized by a tax on refinery emissions, not by a tax on the final uses of the fuel products of the refinery. This remains true even if there is a clear economic and physical linkage between the final use of the refinery products and the emissions from the refinery[7]. Similarly, to internalize damages from, say, oil spills, we should tax the oil and activities that actually put the environment at risk, not the motor fuel that eventually is made from oil.

Unfortunately, the situation is not quite as simple as estimating marginal damages and applying a tax equal to same (if indeed estimating marginal damages is simple). As Baumol and Oates (1988) note:

> The optimal tax level on an externality-generating activity is not equal to the marginal net damage it generates *initially,* but rather to the damage it would cause if the level of the activity had been adjusted to its *optimal* level (p. 160-161; emphasis in original).

They suggest that this optimal tax could be reached iteratively:

> Instead of trying to go directly to the optimal tax policy, as a first approximation, one could base a set of taxes and subsidies on the current damage (benefit) levels. In turn, as outputs and damage levels were modified in response to the present level of taxes, the taxes themselves would be readjusted to correspond to the new damage levels. It might be hoped that this would constitute a convergent iterative process with tax levels affecting outputs and damages, these, in turn, leading to modifications in taxes, and so on (p. 161).

There is no guarantee that this will work, however, and hence no guarantee of really having an optimal taxing policy.

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[7]Note, though, that if there is such a linkage, then it is reasonable to count the environmental damages from refinery emissions as a cost of the various final uses, because the final product uses do, through a chain of events, give rise to the environmental costs of the refinery. Nevertheless, linkages or no, it remains true that the emissions tax should be levied at the refinery stacks. Thus, even though we may say that refinery pollution is a cost of motor-vehicle use, to internalize the damages from the refinery we should tax the refinery and not the motor-vehicle use. Of course, if we do actually tax the refinery, then there no longer will be an external cost to add to the cost of end use -- in this case, the environmental damages from the refinery will be incorporated already in the end-use cost of fuel.

9.1.3 Some additional complications in pricing externalities

There are several other complications in externality pricing. I discuss two more here.

The "piecemeal" problem. First, if an "optimal" taxation-internalization policy is *not* applied universally, to every externality in the economy -- that is, if it is applied "piecemeal," to only a few sectors -- say, to electricity generation by regulated utilities, but not to home use of natural gas or wood -- not only might social welfare not be increased to the optimal level, it might actually be diminished to below the level obtaining before the piece-meal tax was instituted.

Suppose, for example, that before the advent of an externality policy a home can be heated by wood at a private cost of $30/million-Btu, or by electricity from the grid at a private cost of $25/million-Btu. Assume that people care only about money cost, and not about other attributes of heating, and therefore choose grid electricity at $25/million-Btu. Suppose further that grid electricity has an external cost of $7/Million-Btu, and wood use an external cost of $5/Million-Btu. Then, before the externality tax, society pays $25 (private) + $7 (external) = $32/million-Btu for home heating by grid electricity. Now suppose that electricity generators are charged $7/Million-Btu for the marginal environmental damages, and that this cost is passed on to consumers, but that there is no externality tax on wood use. Consumers now face the social cost of electricity, $32/Million-Btu, but only the private supply cost of wood, at $30/million-Btu, and consequently choose wood. But the social cost of wood is $35/Million-Btu -- higher than the social cost of the electricity that was used before the advent of the tax. (Of course, because of the higher price per unit of energy, consumers will use less wood-energy than they did electrical energy.)

In general, the extent to which this "piecemeal" problem might actually reduce social welfare depends on the supply and demand curves for the taxed activity and the substitutes for the activity. In an analysis of a related problem in the electricity-generating sector, Palmer and Dowlatabadi (in Krupnick [1993]) found that if investment decisions for new power plants were based on social cost, but dispatch decisions (about which plants to operate) were based on private cost, the "new source bias effect and the perverse effect on emissions was small" (p. 17). In any event, this sort of problem never arises if all externality-generating activities (or at least all activities within a given economic sphere of potential substitutes) are taxed appropriately.

Externality pricing given existing regulations. A second complication is that if externality pricing is piled on top of existing environmental regulations, such as emissions standards, then under some circumstances the optimal tax might not be equal to marginal residual damages (Burtraw et al., 1993). The relationship of the optimal tax to marginal damages depends on the type of pre-existing regulation (e.g., whether command-and-control or tradable permits), and whether or not prices are set at marginal or average cost. If the existing regulation is an emissions standard, and if prices are approximately equal to marginal cost and there is little opportunity for "bypass" (that is, for substituting an untaxed activity, as in the example above), then generally the optimal externality tax still is equal to marginal damages (Burtraw et al., 1993). I believe, but do not demonstrate, that these conditions characterize the provision and use of motor vehicles. If so, then we may doubt Norman's (1993) claim that emissions regulations internalize damages from motor-vehicle emissions because the per-mile cost of the regulations is about the same as the per-mile cost of the damages. Regardless of the cost of the regulations, the optimal strategy, *given* the regulations, *still* will be to tax the residual damages.

For a additional discussion and analysis of externalities and environmental policy, see Button (1993, 1994), Cropper and Oates (1992), and Baumol and Oates (1988)[8].

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[8]There are of course plenty of methodological difficulties in estimating external costs. One serious difficulty concerns the use of the contingent valuation method, in which people are surveyed to determine their willingness to pay (WTP) for environmental amenities in hypothetical markets. In some cases, the sum of the WTP for several environmental improvements, estimated for example in separate contingent valuation surveys, might exceed the WTP for all of the environmental improvements considered at once. This will be the case if people have something akin to a fixed budget for environmental improvements, and if when faced with a hypothetical question about WTP to pay for a single environmental improvement, do not realize that other environmental improvements might compete for the resources within their limited environmental budget. For example, one survey might find that people are willing to pay $X/year to reduce oil pollution in the ocean, and another independent survey might find that people are willing to pay $Y/year to reduce gasoline pollution in groundwater. It is possible, though, that if people were asked to fund a hypothetical program to reduce the oil pollution and the gasoline pollution simultaneously, they would be willing to pay less than $(X+Y)/year for the same reductions.

More broadly, Hanley (1992) and Stirling (1997) discuss some of the theoretical, ethical, political, and general methodological issues in valuing external costs: equity versus efficiency, risk perception, valuation of non-market goods, ecosystem complexity, the social rate of discount, irreversibilities, treatment of uncertainty, systems boundaries, and more.

9.1.4 The relevance of abatement costs to external costs and damage costs

Some practitioners use the cost of pollution abatement as an estimate of the cost of the damage of pollution -- a practice that economists abhor. This gives rise to the following question: can abatement costs ever "represent" damage costs, or be used in place of damage costs?

This question actually concerns semantics, not theory. It is indisputable that the actual cost of controlling emissions from a particular source is conceptually distinct from the cost of the damage caused by the emissions that remain. The first has to do with the value of the resources employed to reduce pollution; the second has to do with the value of the resources damaged by the remaining emissions. Moreover, the two numbers -- marginal cost of control and marginal cost of damage -- in general will not be the same, unless there is a policy or incentive or intent that tends to make them so. Finally, not only are the two kinds of costs conceptually distinct, they are *additive* in estimates of social cost: the cost of control is part of the private-cost component of social cost, and the cost of damage is part of the external-cost component of social cost.

So far so good. The confusion arises when people use control costs to represent damage costs. On the surface, this representation might appear to be a misunderstanding of the conceptual difference between the actual cost of controlling emissions and the cost of damage from emissions. Although it might well be a conceptual misunderstanding, it need not be. It is a conceptual confusion to believe that control costs are the same thing as, or *necessarily* are equal to, damage costs, marginally or otherwise. But it is *not* a conceptual confusion (although it might be empirically incorrect) to believe that society sets standards so that control costs are roughly equal to damage costs. Indeed, at the social optimum, the marginal cost of abatement will equal the marginal damage from the remaining emissions, so that it certainly is desirable, at least, that estimates of damage costs inform social policy making in regards to abatement costs.

Therefore, if one believes that standards are set so that control costs are roughly equal to damage costs, *and* if the basis of this belief is *not* an explicit analysis of the cost of control and the cost of damages, then it is reasonable to use estimates of control cost to approximate damage costs. (If the basis of the belief in the rough equality of control costs and damage costs is an explicit analysis of both, then obviously one should use the explicit estimates of damage costs directly.) In other words, if one believes that decision makers compare the dollar cost of control with the physical (not monetized) impacts of control and set standards so that at the margin the additional control cost is "worth" -- in a holistic, non-monetized evaluation -- the additional physical impacts (foregone damages), then one can say that control costs are a good approximation of damage costs.

Of course, one can doubt that decision makers set standards so that control costs roughly equal damage costs, and that, even if they did, a qualitative comparison of dollar costs with physical impacts is superior to explicit estimates of the damage.

9.1.5 Nonmonetary versus monetary externalities

This analysis distinguishes monetary from non-monetary externalities. The distinction here is *not* between cost items that "ought" to be valued in dollars and costs that ought not, nor between efficiently and inefficiently priced items, but rather between cost items that are traded in real markets and hence valued directly in dollars, and items that are not.

Although this distinction is not directly relevant to efficiency of resource use, it is relevant to the practical estimation of social cost. Abstractly, the social cost of any item X (tires, roads, disturbance by noise, suffering from asthma caused by air pollution...) is equal to the quantity of X (number of tires, miles of roads, excess decibels of exposure, days of suffering asthma) multiplied by the unit cost of X ($/tire, $/road-mile, $/excess decibel, $/day of suffering). In this analysis, the distinction between "monetary" and "nonmonetary" costs pertains to the estimation of the $/unit part of the calculation of social costs. An item is classified as a "monetary" cost if we can observe or estimate its $/unit cost (or value) directly from market transactions. Thus, because we can observe the $/unit cost of tires, and the $/mile cost of building roads, tires and roads are classified as monetary costs. By contrast, we cannot observe directly the unit cost of noise or air pollution ($/decibel, or $/day of suffering), because noise disturbance and suffering per se are not traded and valued in markets[9].

The distinction is methodologically important because (obviously) it is much more difficult to estimate the $/unit cost of nonmonetary items than of monetary items[10]. To estimate the $/unit costs of (or demand curves for) nonmonetary items there are a variety of techniques, including hedonic-price analysis and stated-preference analysis, but as noted above all of the techniques can be problematic, and as a result the

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[9]However, protective or ameliorative measures, such as ear plugs or asthma medicine, often are valued in markets. Ideally, one would distinguish these as monetary externalities. Moreover, the entire cost of crop loss due to motor-vehicle air pollution is a market cost, and hence should be classified as a monetary externality. Nevertheless, I classify these as non-monetary externalities because the bulk of pollution costs are non-monetary, and it seems more natural to classify all of the costs of pollution in one category. And in any event, the failure to distinguish all monetary costs does not undermine the classification with respect to efficient resource allocation and proper pricing , because from that perspective there is no difference between a monetary externality and a non-monetary externality.

Why then bother to distinguish monetary from non-monetary externalities at all? One reason, explained in the text, is that non-monetary externalities usually are harder to estimate and more uncertain. A second reason is that some public-sector infrastructure and service costs can be considered to be monetary externalities, and hence to straddle the public-sector and the monetary-externality categories. If we do not distinguish monetary from non-monetary externalities, then some of the public infrastructure and service costs, such as fire protection, will straddle the category that includes environmental externalities, such as global warming. This seems too much of a stretch; it is better to separate public-sector costs from environmental externalities by having an intermediate category called "monetary externalities".

[10]Also, monetary costs, being more tangible, might be more significant politically.

social nonmonetary costs of motor-vehicle use often are very uncertain -- typically, much more uncertain than are the monetary costs[11].

9.2 ESTIMATES OF THE EXTERNAL COST OF MOTOR-VEHICLE USE

9.2.1 Pain, suffering, death, and lost nonmarket productivity due to motor-vehicle accidents

In 1991, motor vehicle accidents damaged nearly 30 million motor vehicles, injured nearly 6 million people, and killed 42,000 people. This property damage, injury, and death cost society several hundred billion dollars in medical expenses, lost productivity, vehicle repair and replacement, pain and suffering, and other costs. In the entire analysis of the social cost of motor-vehicle use, only travel time is more costly.

In Report #19, I derive expressions for the total cost and the external cost of motor-vehicle accidents as a function of vehicle miles of travel, the rate of accidents, and the cost per accident. I begin with a simple expression that equates the total social cost of accidents to the product of the number of persons injured (or killed), or vehicles damaged, and the social cost per person injured or vehicle damaged. Then, I express the number of accidents as a function of vehicle miles of travel (VMT). The first derivative of this total social-cost function is the marginal social-cost function, which can be used to estimate what I call the *potential* external cost: the difference between the marginal social cost and the marginal private cost. The actual external cost is the potential external cost less any user payments, such as liability insurance premiums.

With functions that distinguish external from "internal" (private, or personal) costs, and cost data that distinguish monetary from nonmonetary costs, I can disaggregate the total accident cost into the four categories of accident costs in this social-cost analysis:

i) *personal (or private) nonmonetary* costs, such as pain and suffering due to injuries from accidents that are not externalities (for example, if a person falls asleep and runs into a tree and injures herself, the pain and suffering from the injury is a personal or private nonmonetary cost)

ii) *private monetary (or priced)* costs, such as the cost of repairing vehicles damaged in accidents that are *not* externalities, or the cost of liability insurance against damages inflicted on others;

iii) *external monetary* costs, such as vehicle repair costs inflicted by uninsured motorists; and

iv) *external nonmonetary* costs, such as pain and suffering and lost non-market production inflicted by others and not covered by user payments.

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[11] Of course, some monetary costs also are difficult to estimate and very uncertain. An example is the GNP loss due to a sudden change in the price of oil.

I distinguish external from private costs because, as discussed above, the economically efficient policy is to price the externality but do nothing about the privately incurred costs, other than keep people informed of the risks they face. I distinguish monetary from nonmonetary costs because the latter are much more difficult to estimate, and hence considerably more uncertain. Also, I distinguish accidents involving non-motorists, accidents involving single motor vehicles, and accidents involving two or more vehicles.

Externalities in motor-vehicle accidents. Person A imposes an external accident cost on person B if A causes accidental harm to B but does not pay the price of the harm, and if the harm would not have occurred had A not driven[12]. If A pays the price of his expected harm (say, through liability insurance), then there is no externality, because A has properly accounted for the expected cost of his actions. If however we have determined that the actions of A have caused a harm, to B, that would not have otherwise occurred, but we have not yet determined whether A has paid the price of the harm, the cost of the harm is, to this point, just a "potential" externality.

From our definition of externality, we can see that an action by A gives rise to a potential external cost if the total cost of the action to society exceeds the cost to A himself -- in other words, if the marginal social cost exceeds the marginal private cost. Just as a the additional driver who slows everybody else thereby causes a congestion externality, so the additional driver who increases everybody else's expected accident cost thereby causes an a potential accident externality. (Note again, "potential," because we have not yet accounted for user payments.) Thus, we estimate the potential marginal external cost as the difference between the marginal social cost and the marginal private cost, where the cost per accident and the frequency of different kinds of accidents are estimated as functions of VMT. The actual external accident cost is equal to the potential external cost less any user payments that internalize damages.

Formally, to estimate the external nonmonetary cost of motor-vehicle accidents, I multiply the number of injuries of various types, and the number of fatalities, and the

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[12]Note that even if B fully accounts for the risk to himself imposed by A, the risk still is an external cost if A has not accounted (paid) for it. Put another way, efficiency demands that risks be accounted for fully by *both* the imposer and the imposee. This is not double counting, any more than it is double counting to require that both the polluter and the pollutee fully consider the costs to the pollutee. We get into trouble here only if victims are compensated by perpetrators, in any situation: pollution, accidents, and so on. As discussed in the text, compensation invites inefficient tolerance of the bad, because it removes the incentive to take optimal defensive measures. (Note that pricing without compensation would have to be run or at least controlled by the government, because otherwise private insurance companies would gain revenues from the liability premium but would not have to pay out any compensation.)

Similarly, one should not be misled because nearly all of the pain and suffering costs of motor-vehicle accidents are "internal" to the class of motor-vehicle users as a whole. This "internalization" by the class is not relevant, because individual drivers, not classes, make decisions. Efficiency conditions (MC = P = MV) must be satisfied at each decision, and if an individual decision maker does not account for the pain and suffering that he might cause (because, in practice, he is not financially liable for the full expected amount of pain and suffering that he might inflict), then that expected pain and suffering is an externality of motor-vehicle use.

number of vehicles involved in property-damage-only (PDO) crashes, by the nonmonetary cost per injury or fatality or vehicle, and by the fraction of the cost that is a "potential" externality, and then deduct priced costs:

$$ENM = \left(\sum_i NM_i \cdot IO_i \cdot OFF_i \cdot PEXT_i \right) - PPENM \qquad \textbf{[9-1a]}$$

$$NM_i = \sum_c NM_{i,c} \cdot W_c \qquad \textbf{[9-1b]}$$

where:

ENM = total external nonmonetary cost of motor-vehicle accidents ($)

NM_i = Nonmonetary cost per injury of type i ($/injury or vehicle)

IO_i = number of persons with MAIS injury type i, or the number of fatalities, or the number of vehicles involved in PDO crashes, on public roads (Report #19; based on Miller [1997] and Blincoe {1996])

OFF_i = factor to account for accidents off the road or on private roads, and for non-collision injuries or deaths (e.g., from falling down while getting into car) (Report #19)

$PEXT_i$ = of total MAIS injuries, or fatalities, or PDO vehicles, the fraction that is a potential externality (see Report #19, and the brief discussion below)

PPENM = potential external non-monetary costs that actually are priced to those responsible, and hence are not actual nonmarket externalities, but rather private monetary [insurance] costs (see Report #19)

$NM_{i,c}$ = the nonmonetary cost type c per MAIS injury, or fatality, or PDO vehicle (Report #19; based on Miller [1997] and Blincoe {1996])

W_c = the fraction of NM_{ic} that is not counted elsewhere in the analysis (Report #19)

subscript i = the accident classifier, representing one of six types of injuries (MAIS 0 to MAIS 5; see section 9.2.14), or fatalities, or vehicles involved in property-damage-only (PDO) crashes

subscript c = the kinds of nonmonetary costs: pain and suffering, and lost nonmarket productivity (household productivity)

Note that equation 9-1 is a condensation of the actual method used, which as mentioned above specifies total cost functions, derives marginal cost functions, and estimates potential external costs formally as the difference between marginal social cost and marginal private cost (which is assumed to equal average cost). This is done for three different categories of accidents (nonmotorist, single-vehicle, and two-or-more vehicle), as well as for the different accident severity classes.

Accident costs: what fraction are a potential external cost? My analysis, based on recent work Newberry (1988), Janson (1994), Elvik (1994), Persson and Ödegaard

(1995), Mayeres et al. (1996), the Transportation Research Board (1996), and others, and presented in Report #19, finds that considerably less than half of accident costs are potential externalities. This is a fair bit lower than the external cost fraciton estimated by Cohen (1994) and even Elvik (1994). Cohen (1994) reports that an analysis of data on insurance costs and fatal accidents for motor carriers suggests that 67% of pain and suffering costs are externalities. Elvik (1994) uses a variety of data and inferences to estimate that, in Norway, the external costs of traffic injury are only about 40% of the total cost of traffic injury. Evans (1994) remarks that "rough calculations suggest that safety externalities are at least as important as congestion, though their distribution in space and time is very different" (p. 5).

How much pain and suffering or lost non-market productivity is priced (compensated)? In order to estimate the actual externality, we need to know how much drivers pay, either out of pocket, or through liability insurance, towards nonmarket costs (pain and suffering and lost nonmonetary productivity) that they inflict on others as potential externalities. However, as Verhoef (1994) notes, this is difficult to estimate.

The difficulty here is two-fold: there are not separate liability premiums for nonmonetary as opposed to monetary damages, and, what's worse for my purposes, the premiums are not levied according to the economic definition of an external cost. There are, however, data on compensation for nonmonetary costs of motor-vehicle accidents. Thereefore, first, I estimate the amount of compensation received for nonmonetary costs of motor-vehicle accidents, as reported by Hensler et al. (1991) for 1988. Then, on the basis of data in Marowitz (1991) and the Bureau of the Census *Statistical Abstract of the United States* (1992, 1996), I estimate how much of the compensation was in effect a price of motor-vehicle use. The analysis is documented in Report #5.

Note that, in principle, we also should determine how much of the compensation was for potential external costs, as opposed to personal nonmarket costs. Because compensation is awarded on the basis of legal liability, rather than on the basis of potential externalities, a person might be compensated for damages that, although legally attributable to another party, are not a potential external cost of the actions of the other party. (See the discussion in Report #19). Unfortunately, I have no basis for making this determination, and so simply assume that all of the compensation is for potential external costs. Note though, that this is merely an accounting problem: it affects only the apportioning of total nonmarket costs between "personal nonmarket" and "external".

Results of the analysis. Table 9-9 shows the estimated total external costs of motor-vehicle accidents.

9.2.2 Travel delay, imposed by other drivers (and including delay due to accidents), that displaces unpaid activities

When a person considers taking a trip, he of course considers how much time the trip will take him, but usually he does not consider how his trip might delay others. This unaccounted-for delay imposed on others is an external cost. The imposed delay can displace paid work (in which case we classify the time cost as a monetary externality, and estimate it in Report #8), or unpaid activities (in which case we classify the time cost as a non-monetary externality and estimate it here). (We classify travel time that is not imposed delay as either a personal non-monetary cost [Report #4], or with motor-vehicle goods and services priced in the private sector [Report #5]). The nonmonetary travel delay externality is one of the larger items in our social-cost accounting.

The external non-monetary time cost of travel is estimated as a function of vehicle occupancy, average speed, and other factors:

$$TTCenm = PHTd \cdot \left(\frac{1}{Oc} + \left(1 - \frac{1}{Oc}\right) \cdot Pa\right) \cdot \left(Fnm,dr \cdot Cnm,ref \cdot \left(\frac{Sref}{\frac{Snd}{R}}\right)^{Bo} + Ch,dr,ref \cdot \left(\frac{Sref}{\frac{Snd}{R}}\right)^{Bh} \right)$$

[9-2]

where:

TTCenm = the external, non-monetary travel-time cost (10^9 1991$)

PHTd = person-hours of travel delay (the travel-time externality) (10^9 person-hours of delay)

Oc = average vehicle occupancy (persons/vehicle)

Pa = the ratio of the cost of passenger time to the cost of driver time

Fnm,dr = the fraction of travel time that displaces non-monetary activities rather than paid activities, for drivers

Cnm,ref = The cost of the foregone non-monetary activities, at the reference speed Sref ($/person-hour; estimated as a function of trip purpose and income class)

Ch,dr,ref = the pure hedonic cost of travel, for drivers, at the reference speed Sref ($/person-hour)

Sref = the reference speed, with respect to which the speed-dependence of Cu and Ch are estimated (assumed to be 30 mph)

Snd = the average speed when there is no delay (mph)

R = over the miles subject to delay, the ratio of average free-flow speed had there been no delay to the average speed given the actual delay

Bo = exponent that determines the dependence of opportunity cost Cnm on average vehicle speed when there is no delay (assumed to be 0.15)

Bh = exponent that determines the dependence of hedonic cost Ch on average vehicle speed when there is no delay (assumed to be 0.75)

All parameters are estimated in Report #4. Note that we disaggregate the total hourly value of travel time, which is the willingness to pay to do something other than drive or sit in a car, into two components: an opportunity-cost component, and a hedonic component (Hensher, 1997). The opportunity cost is the value of activities foregone while in the car. While in a car, a person may forego paid work, unpaid activities such as leisure, or nothing at all (if, for example, the person is able to work while in the car). The hedonic cost is the pure utility or disutility of the driving experience itself. The hedonic cost is determined by several factors, including comfort, safety, privacy, available space, amenities, and the amount of effort and attention required to control a vehicle. If one actually likes driving, then this hedonic cost is negative.

Note too that I assume that the opportunity cost per hour declines with increasing average speed. The relatively weak basis for this is given in Report #4.

9.2.3 The cost of the health effects of air pollution from motor vehicles (Report #11)

Motor vehicles and their related emission sources, such as petroleum refineries, emit many different kinds of air pollutants, which affect human health in a variety of ways. These health effects create a large economic cost to society. In Report #11, we estimate the social cost of many of the health effects of motor-vehicle air pollution.

The relationship between changes in emissions related to motor-vehicle use and changes in health welfare (measured in dollars) can be modeled in three steps:

1) relate changes in emissions to changes in air quality;
2) relate changes in air quality to changes in physical health effects; and
3) relate changes in physical health effects to changes in economic welfare.

We have made a detailed model of this sort to estimate the cost of the health effects of motor-vehicle air pollution.

We estimate the human-health cost of motor-vehicle air pollution in the entire U.S., in urban areas of the U.S., in rural areas of the U.S., and in 11 major metropolitan statistical areas (MSAs): Boston, Denver, Houston, Los Angeles, Minneapolis, New York, Philadelphia, Phoenix, St. Louis, Spokane, and Washington D. C. We consider six types of motor vehicles: light-duty gasoline and diesel vehicles, light-duty gasoline and diesel trucks, and heavy-duty gasoline and diesel trucks. We estimate the number and type of health effects, and the monetized value of these effects, including total dollar costs, dollar costs per vehicle-mile of travel, and dollar costs per kg of pollutant emitted. Finally, we include an analysis of the three main sources of the costs: direct emissions from motor vehicles, emissions of road-dust particulate matter, and "upstream" emissions from gasoline stations, refineries, vehicle manufacturing, and so on.

Emissions and air quality. We estimate the *status quo* air quality in 1990, and then estimate the effect of reducing motor-vehicle-related emissions by 10% and 100%, and reducing all anthropogenic emissions by 100%. We represent the *status quo* with measurements of actual ambient air quality at air-quality monitoring sites (EPA, 1993d). To estimate air quality without 10% or 100% of motor-vehicle related emissions, we use a simple model of emissions, dispersion, and atmospheric chemistry, developed in Report #16 and summarized as follows:

$$PI_{P,c}* = C_{P'\rightarrow P}\begin{pmatrix} \sum_i \left(E_{P1',i,c} \cdot D_{P1',i,c} + E_{P1',i,oc} \cdot D_{P1',i,oc}\right) \\ \sum_i \left(E_{P2',i,c} \cdot D_{P2',i,c} + E_{P2',i,oc} \cdot D_{P2',i,oc}\right) \\ \ldots \end{pmatrix} \qquad \textbf{[9-3a]}$$

$$PP_{P,c}* = C_{P'\rightarrow P}\begin{pmatrix} \sum_i \left(E_{P1'\wedge,i,c} \cdot D_{P1',i,c} + E_{P1'\wedge,i,oc} \cdot D_{P1',i,oc}\right) \\ \sum_i \left(E_{P2'\wedge,i,c} \cdot D_{P2',i,c} + E_{P2'\wedge,i,oc} \cdot D_{P2',i,oc}\right) \\ \ldots \end{pmatrix} \qquad \textbf{[9-3b]}$$

where:

$$E_{P',i,oc} = \sum_{o \in R_c} E_{P',i,o}$$

$PI_{p,c}*$ = the modeled level of total ambient pollution P "received" or formed at air-quality monitors in county C, in a year, given the baseline emissions

$PP_{p,c}*$ = the modeled level of total ambient pollution P "received" or formed at air-quality monitors in county C, in a year, after the change in emissions

$C_{p'\text{-->}p}$ = the chemical transformation of emissions of precursor pollutants P′ (P1′, P2′,...) to ambient pollutant P (discussed below; this transformation function is assumed to be the same in every county, and to be independent of the source of the emissions)

$E_{p1',i,c}$, $E_{p2',i,c}$ ... = yearly baseline emissions of precursor pollutants P1′, P2′... from emissions source i in county C

$E_{p1',i,oc}$, $E_{p2',i,oc}$ ... = yearly baseline emissions of precursor pollutants P1′, P2′... from emissions source i in all counties except C in AQCR R

$D_{p1',i,c}$, $D_{p2',i,c}$ = the fraction of emissions of precursor pollutants P1′, P2′... from source i in county C that reaches the ambient air-quality monitor in county C

$D_{p1',i,oc}$, $D_{p2',i,oc}$ = the fraction of emissions of precursor pollutants P1′, P2′..., from source i in all counties except C in AQCR R, that reaches the ambient air-quality monitor in county C

$E_{p1'^,i,c}$, $E_{p2'^,i,c}$ ... = yearly emissions of precursor pollutants P1′, P2′... from source i in county C, after the change in emissions

$E_{p1'^,i,oc}$, $E_{p2'^,i,oc}$ ... = yearly emissions of precursor pollutants P1′, P2′... from source i in all counties except C in AQCR R, after the change in emissions

$E_{p',i,o}$ = emissions of pollutant P′ from source i in county O in AQCR R_c (for simplicity, we leave the notation for P′ general, and do not write out separate equations for P1′, P2′, P1′^, and P2′^)

subscript P = the ambient pollutants, measured at the ambient air-quality monitors: (CO), ozone (O_3), nitrogen oxides (NO_x), total suspended particulate matter (TSP), particulate matter less than 10 microns in aerodynamic diameter (PM_{10}), and particulate matter less than 2.5 microns ($PM_{2.5}$)

subscript P′ = the emitted pollutants: CO′ (-> CO), PM_{10}′ (-> PM_{10}), $PM_{2.5}$′ (-> $PM_{2.5}$), NO_x′ (-> NO_2, O_3, PM_{10}, $PM_{2.5}$); volatile organic compounds (VOCs′; -> O_3, $PM_{2.5}$), SO_2′ (-> PM_{10}, $PM_{2.5}$), ammonia (NH_3′ -> PM_{10}, $PM_{2.5}$)

subscript C = the county of interest (i.e., the county for which air quality and the cost of air pollution are estimated)

subscript R_c = the AQCR that contains county C

subscript OC = all counties other than county C in AQCR R_c

We estimate the effects of a specific, "marginal" change in pollution: the difference between actual pollution and, what pollution would have been had there been either a 10% or a 100% reduction in motor vehicle-related emissions. We emphasize two points here. First, it is useful to consider a marginal change because the formation of ambient pollution from emissions is a nonlinear process, and some of the effects of ambient pollution on people's health are modeled as nonlinear. Second, when we say "motor vehicle-related," we include all "upstream" or "indirect" emissions associated with motor-vehicle use, as well as tailpipe emissions from vehicles: emissions from petroleum refineries, road dust, emissions from the servicing of motor vehicles, emissions from road construction, and so on.

I emphasize that our air quality modeling is crude, and considerably behind the state of the art. Douglas et al. (1998) document their recent application of more sophisticated models, in a project to estimate the social costs of highway transportation in the year 2000: a 3-dimensional photochemical model, the Urban Airshed Model, to estimate ozone air quality, and an advanced Eulerian regional model, called REMSAD, to simulate short-and long-distance transport of particulates and particulate precursors.

For the purpose of modeling air quality levels as a function of emissions, these sophisticated models are vastly superior to the simplistic dispersion and chemistry models. However, if one is modeling not air quality per se, but rather the relative contribution of different sources to known (measured) air quality, then the shortcomings of the simplistic models compared to the advanced models may be tolerable.

Air pollution and health effects. We estimate the health cost of four "criteria" pollutants (carbon monoxide, nitrogen dioxide, ozone, and particulate matter) and six "toxic" air pollutants (formaldehyde, acetaldehyde, benzene, 1,3-butadiene, gasoline particulates, and diesel particulates). The estimation methods for the criteria pollutants are different than the methods for the toxic air pollutants.

Criteria pollutants. We reviewed hundreds of clinical, animal, and epidemiological studies of the health effects of various pollutants, and constructed exposure-response functions for each criteria pollutant (ozone, carbon monoxide, etc.) and each of a variety of health effects (for example, asthma, or headaches). These functions relate the change in health effects to the change in exposure. We have developed mortality-risk estimates for those pollutants, such as fine particles, which according to some studies are associated with mortality. For most pollutants and health effects, we have established upper and lower-bound estimates of the effects of exposure (Table 11.1-1).

In general, an exposure-response function has the following form:

$$\Delta E = f(\Delta P, O) = f(PI, PP, O) \qquad \textbf{[9-4]}$$

where:

ΔE = the change in the effect of interest (in this case, human health)
ΔP = the change in ambient air pollution
O = other variables (such as population in the county, or the incidence rate of a health problem or cause of death; see Chapter 11.3 of Report #11)
PI = the initial pollution level (estimated from data on actual ambient air quality in counties in the U.S.; see Chapter 11.2 of Report #11)
PP = the pollution level after the change in pollution -- in our analysis, the level had there been no motor-vehicle-related emissions (detailed in Report #16).

As a specific example, here is the function that estimates the number of acute cardiovascular deaths caused by PM_{10} in each county (derived in Chapter 3 of Report #11):

$$\Delta CVD = CVD_{PI} \cdot \left(e^{(PP-PI)\cdot\beta_{PM}} - 1\right) \qquad \textbf{[9-5]}$$

where:

Δ CVD = change in the daily number of cardiovascular deaths
CVD_{PI} = county daily average number of cardiovascular deaths at the initial pollution level PI (U.S. Department of Health and Human Services, 1991)
PI = the initial (1990) daily average PM_{10} level ($\mu g/m^3$)
PP = the daily average PM_{10} level ($\mu g/m^3$) after the change in pollution
β_{PM} = 0.00179, PM_{10} Poisson regression coefficient (Pope et al., 1992).

Our estimate of the health effects of particulate matter, which is by far the most damaging pollutant, accounts for the likelihood that smaller particles are more damaging than larger particles, that geological material is less damaging than combustion material, and that particulate-matter emission inventories are seriously mis-estimated.

Toxic air pollutants. Whereas the cost of the criteria pollutants is estimated on the basis of human epidemiological studies and ambient air-quality data, the cost of toxic air pollutants is estimated on the basis of unit-risk values and exposure to pollution in micro-environments. Unit-risk functions relate the probability of getting a particular type of cancer (e.g., leukemia) to the amount of exposure to a particular toxic air pollutant (e.g., benzene). Details are given in Chapter 11.6 of Report #11.

Valuation of health effects. In the last step of the damage-function method, we estimate the economic value of the estimated health effects. Our estimates of the dollar value of health effects are derived from studies of the value of lost work days, of restricted activity, of tolerating certain symptoms of illness, and so on. When we estimate the value of life (VOL), which is the most important valuation parameter in the analysis, we distinguish future deaths from current deaths, and deaths that would have occurred soon anyway even if there were no pollution from deaths that would not have. We also assume that air pollution mainly kills the elderly, and that the VOL of the elderly is a bit less than the VOL of the middle aged working males for whom VOLs typically are derived. However, some recent work implies that the VOL for the elderly might be an order of magnitude lower than the VOL for young people (Cropper et al., 1994; Johannesson and Joahnsson, 1999).

The total health cost then is equal to the change in the effect of interest (ΔE in equation 9-4; e.g., number of deaths due to motor-vehicle particulate air pollution) multiplied by the dollar value per effect (e.g., the value of life).

Summary of results. The results of our analysis of the health cost of motor-vehicle air pollution are summarized in Tables 9-1a to 9-1d and 9-9. Tables 9-1a to 9-1d to show the $ cost of health effects per kg of each pollutant emitted, at the level of a 10% reduction in motor-vehicle use, for three different geographic regions. Results are shown cumulatively for a 10% reduction in emissions from motor vehicles only, a 10% reduction in emissions from motor vehicles and upstream sources, a 10% reduction in emissions from motor vehicles, upstream sources, and paved roads, and finally a 10%

reduction in emissions from motor vehicles, upstream sources, paved roads, and unpaved roads.

The $/kg values arguably are the most useful results, because damages per kilogram emitted, unlike damages per mile of vehicle travel, or total dollar damages, are not affected by the uncertainty in the emissions inventory. The $/kg figures can be applied to any assumed vehicular emission rate or emissions inventory. For example, one can use them to estimate the cost of future low-emitting vehicles or alternative-fuel vehicles that have emission rates different from the present national-average rates. Note, though, that the $/kg values *are* proportional to the *exposed* population: if one expects the exposed population to increase by 10% over 1990 levels, then one should increase the pertinent $/kg values by 10%. Similarly, the $/kg estimates are proportional to the assumed value of health effects. They also depend somewhat on the total change in pollution or emissions being considered, because some health effects are non-linearly related to pollution levels. However, the dependency is not strong: most of the major costs either vary linearly with pollution levels (in which case the $/kg cost is independent of the pollution level), or else nearly linearly.

The $/kg factors for different aggregated emission levels (e.g., motor-vehicles only, versus motor-vehicles plus related upstream emissions) incorporate differences in the size of the population exposed, the ratio of effective exposure to emissions, the potency of particles by size class, and the potency of particles by composition. Thus, we see in Tables 9-1a to 9-1d that the $/kg cost of particulate emissions directly from motor vehicles vastly exceeds the $/kg cost of particulates from motor vehicles + upstream + road dust, because motor vehicle PM is more potent than is road-dust PM, and because more people are exposed to motor-vehicle PM than to road-dust PM.

Table 9-9 shows the total cost of human mortality and morbidity due to particulate emissions from motor vehicles, other pollutants from motor vehicles, upstream emissions related to motor-vehicle use, and road-dust emissions, in billions of dollars.

<u>Discussion</u>. The most important result we found is the large cost of particulate matter pollution, and the potentially large contribution of motor vehicles to ambient particulate levels (Chapter 11.7 of Report #11). Generally, combustion emissions of particulate matter and precursors to particulate matter cause the greatest health costs, by far.

Particulates appear to cause a number of respiratory ailments, including chronic illness and mortality. Motor vehicles contribute the smaller, more dangerous particulates directly from tailpipe emissions and indirectly from the large amounts of "precursor" gases that they give off such as nitrogen dioxide. Motor vehicles also emit large amounts of fairly coarse soil-based particulates from road dust -- dust kicked up into the atmosphere from moving vehicles.

Carbon monoxide, nitrogen dioxide, ozone, sulfur dioxide and toxics appear to have much smaller effects than particulates. Aside from their contribution to particulate formation, emissions of nitrogen dioxide, sulfur dioxide and volatile organic compounds are relatively unimportant. Interestingly, in the cost ranking, ozone, which is formed from the interaction of nitrogen oxides and volatile organic carbon, is nearly in last place, well behind particulates, and less damaging even than carbon monoxide and nitrogen oxides. In part this might be due to our inability to capture all of the effects of ozone. Nevertheless, it may be that air pollution policy has focused too heavily on ozone control and not enough on particulate control.

Uncertainty. Of course, at every stage of the modeling process there is considerable uncertainty, which generally we represent with lower and upper-bound estimates. For several of the emission estimates, the difference between the lower and upper-bound estimates is a roughly a factor of two, and for most of the valuation functions or parameters, the difference is at least a factor of four. Our air quality modeling is simplistic. With the dose-response functions, we have several kinds of problems: we do not have functions for every plausible health effect; in only a few cases do we distinguish susceptible sub-populations; we cannot be sure that the explanatory pollutant variables really are the cause of the statistically associated health effects; and we can only guess at the importance of the size or composition of particulate matter. Turning to valuation, we do not precisely who air pollution harms, and in the case of mortality, how the VOL varies with the age of the victim. (As noted above, recent studies suggest that the VOL for the elderly is an order of magnitude lower than the VOL for younger people.) All told, the uncertainty compounds into an order-of-magnitude difference between the low and the high estimates of total cost (Table 9-9).

For a recent review of the literature on the air pollution damages of transportation, see Krupnick et al. (1997). See also the recent methodological reviews by Krupnick (1993) and Cifuentes and Lave (1993). For a recent estimate of air-pollution damages in Los Angeles, see Small and Kazimi (1995).

9.2.4 The cost of reduced visibility due to particulate air pollution from motor vehicles (Report #13)

Introduction. Particles and gases in the atmosphere scatter and absorb light, and thereby reduce visibility (Watson and Chow, 1994; Richards et al., 1990). Although natural sources of particles, such as volcanoes, can significantly degrade visibility, it generally is true that "when visibility is poor...most particles are found to be of human origin, from sources such as power plants, vehicle exhaust, biomass burning, suspended dust, and industrial activities" (Watson and Chow, 1994, p.244). Poor visibility diminishes the enjoyment of scenic vistas and makes travel hazardous. Statistical analyses of property values (discussed below) reveal that people are willing to pay a premium for houses in areas with good visibility and air quality.

The particles that are most efficient at scattering light are about the same size as the wavelength of visible light -- about 0.5 μm. Most particles emitted by the combustion of diesel fuel, and some particles of re-entrained road dust, are between 0.1 and 1.0 μm. Hence, the use of motor vehicles potentially is a significant cause of visibility degradation.

General Estimation Methods. There are two ways to estimate the cost of impaired outdoor visibility: contingent valuation (CV), and hedonic price analysis

(HPA). The CV and HPA methods have complementary strengths and weaknesses (see Report #13 and Chestnut and Rowe, 1990a [updated but abridged in Chestnut and Dennis, 1997], for further discussion.) With CV, researchers survey people and ask them to make explicit, but hypothetical, tradeoffs between visibility and dollars or things with a known dollar value. The main strength of CV is that it is explicit: the item to be valued (in our case, visibility) is identified and described and "marketed" explicitly. However, the obvious and potentially grave weakness of CV is that the valuation is hypothetical, and therefore reliable only insofar as people respond realistically to the hypothetical market.

In HPA, researchers analyze the value of visibility that is implicit in the prices that people pay for houses in regions that have different average annual levels of visibility or air quality. The strength of hedonic price analysis is that it is based on real, "revealed" behavior in the market place. However, the individual items being valued, such as air quality, are not actually marketed explicitly as separate items, but rather are marketed implicitly, as part of a bundle of many attributes. This makes it difficult to know what aspects of visibility or air quality people are valuing (e.g., the aesthetic component only, or the associated health effects as well).

In this report, we will use the hedonic model Smith and Huang (1995) to estimate the willingness to pay (WTP) for improved visibility.

The model. Smith and Huang (1995) perform a meta-analysis of hedonic price analyses of the marginal willingness-to-pay (MWTP) to reduce levels of total suspended particulate (TSP). They reviewed over 50 studies developed between 1967 and 1988, 37 of which had some empirical estimates involving hedonic price functions with some measure of air pollution.

Because the meta-analysis synthesizes many different studies from many different regions, it is as good a basis as any for estimating national damages. Another advantage for us of the Smith and Huang (1995) model is that the independent pollution variable in their analysis, TSP, not only is highly correlated with visibility, it actually is the main physical cause of reduced visibility.

We begin with their minimum-absolute-deviation (MAD) demand equation, which estimates the MWTP per household, in 1982-1984$, per $\mu g/m^3$ of TSP, as a function of the per-capita income and the TSP level:

$$V_{83} = \alpha + \beta 1 \cdot P + \beta 2 \cdot Y_{83} \qquad \textbf{[9-6]}$$

where:

V_{83} = the shadow price of visibility: the change in the asset value of the house per unit of pollution ($/[$\mu g/m^3$]), at TSP level T, in 1982-1984 prices (we take 1983 as the base year)

α = intercept (-49.31 in simple MAD model)

$\beta 1$ = coefficient on TSP (-0.23 in the simple MAD model)
$\beta 2$ = coefficient on income (0.01) in the simple MAD model)
P = total suspended particulates (in micrograms per cubic meter)
Y_{83} = average per-capita income in 1982-1984 (we take 1983 as the base year)

We wish to use 1990 data on income and visibility. To do this, we estimate Y_{83} as 1990 income in 1983 dollars, input 1990 TSP levels for P, calculate the resultant V_{83}, which is in 1983 dollars, and finally convert the 1983 $ results to 1991 $.

We treat equation 9-6 as the household demand function for TSP reductions. To calculate how much households in the U.S. are willing to pay for an improvement in TSP, we integrate the household demand function between the two TSP levels, and multiply by all households in the U.S. We estimate the cost of all anthropogenic visibility pollution, and the cost of motor-vehicle visibility pollution. We end up with:

$$VT = \sum_c \left(H_c \cdot \left(\alpha \cdot K1 \cdot \left(PI_c - PP_c \right) + \frac{\beta 1}{2} \cdot K1 \cdot \left(PI_c^{\,2} - PP_c^{\,2} \right) + \frac{K1}{K2} \cdot \beta 2 \cdot Y_c \cdot \left(PI_c - PP_c \right) \right) \right)$$

[9-7]

where:

α, $\beta 1$, $\beta 2$ are as defined above
VT = the total amount extra that all households in the U. S. would have been willing to pay for their homes, if they had bought their homes outright in 1991, if TSP in each county were at the level represented by PP instead of the level represented by PI
H_c = the number of households in county c in the U. S. in 1990 (Bureau of the Census, 1994)
PP_c = what the TSP level in county c would have been in 1990 given no anthropogenic (case I) or motor-vehicle-related (case II) emissions (discussed below)
PI_c = the actual TSP level in county c in 1990 (discussed below)
Y_c = average annual per capita income in county c in 1990 ($/year) (Bureau of the Census, 1994)
K1 = Price deflator to estimate 1991 WTP given 1983 prices (GNP implicit price deflator = 1.322)
K2 = Price deflator to estimate 1990 income given 1983 prices (GNP implicit price deflator = 1.264)
subscript c = counties in the U.S.

Equation 9-7 is our cost model. Note that the estimated total willingness to pay, VT, represents a one-time payment for a commodity (a home) that lasts many years.

Thus, to calculate an annual WTP, the one-time total VT must be amortized, or annualized, over the economic life of the home.

Note too that equation 9-7 uses TSP, not visibility, as an explanatory variable. However, not only are TSP and visibility highly correlated, they in fact are physically related -- as mentioned above, particulate matter scatters light and thereby reduces visibility -- which means that we can estimate how TSP affects visibility. The real difficulty will be to determine how much of the WTP to reduce TSP is WTP for visibility per se, as opposed to WTP to reduce the health and other effects of air pollution. We will analyze this below.

The portion of the total WTP that is for visibility per se. Hedonic price analyses relate differences in house values to differences in some measure of air quality. Given any such estimated relationship, and keeping in mind that our objective is to estimate the cost of visibility degradation, we are faced with two questions: First, is the TSP measure of air quality in the meta-analysis model of Smith and Huang (1995) the right one? Second, what is it about air quality that people value?

<u>The right air-quality measure?</u> Ideally, one would use as an explanatory variable the measure of air quality that people actually have in mind when they buy a house. To the extent that the air-quality explanatory variable in a hedonic model is *not* correlated with the real air-quality variables in people's minds, the model will mis-estimate the relationship between housing value and air quality

Most likely, prospective home buyers do not actually consult statistics from air-quality monitors, but rather judge air quality on the basis of whether or not the air appears polluted, and what people and the media say about the local air pollution. If this is so, then visual range, or some close proxy, probably represents reasonably well the "air quality" as perceived and evaluated by people. Because TSP is closely correlated with visibility, we assume that it adequately represents the air quality that people actually are evaluating.

<u>What do people value about good air quality?</u> Even if we have the right measure of air quality, we still need to identify the "components" of air quality that people care about. When people pay more for a house in an area with cleaner air, what benefits do they think that they are buying? Better health? Reduced soiling of clothes and materials? Or just better visibility? The question is important to us because our goal here is to measure the value of visibility or aesthetics per se.

Most likely, the visibility benefit is not the bulk of the total perceived air-quality benefit. Smith and Huang (1995) argue that the "hedonic models....reflect aesthetics, materials, and soiling effects, and, to some degree, perhaps perceived health effects, although the latter may well be incomplete" (p. 223). In support of this, Brookshire et al. (1979, 1982) found that, of the estimated willingness-to-pay for improved air quality in the South Coast Air Basin, about 34% was for improved aesthetics (which we would call visibility per se), and the rest for improved health. In a survey study of WTP for visibility, Loehman et al. (1994) found that the value of visibility was 10% to 40% of the total health+visibility value of air quality. In their survey of WTP to improve air quality in the eastern U. S., McClelland et al. (1991) found that the value of improving visibility per se was 19% of the total health+visibility+soiling+ "other" value of air quality (also cited in Chestnut and Dennis, 1997).

Given this, we assume that value of visibility per se constitutes 15% to 35% of the total value of "air quality" estimated by the Smith and Huang (1995) hedonic model.

The value of visibility outside of the local housing market. Another shortcoming of the hedonic-price approach is that it captures the value of air quality, or visibility, in housing markets only; it does not capture any visibility value in other markets (Chestnut and Dennis, 1997; Cropper and Oates, 1992; Chestnut and Rowe, 1990a).

When people assess visibility when they shop for a home, they assess the differences in the "visibility experiences" that will result from choosing one home over another. For example, they certainly will compare visibility in and around the candidate houses, because those local visibility experiences will depend on which home they buy. But buyers will not consider visibility in areas that they will visit (or, more generally, that they will care about) *regardless* of which home they buy. Thus, to the extent that persons care about visibility outside of their home region (or housing market), the hedonic-price estimate, used by itself, will underestimate the total value of visibility *everywhere*.

It seems clear to us that people and environmental regulators care about visibility in wildernesses, National Parks, scenic areas, and urban areas outside of their home region or housing market. However, it is difficult to find data that one can use to extrapolate the hedonic results to include visibility values not captured in housing markets. On the basis of the findings and judgement in Chestnut and Rowe (1990a) and Chestnut and Dennis (1997) (see Report #13), we judge that visibility value not captured by housing markets is 40% to 70% of the value estimated by the hedonic model. Thus, we multiply the household results by 1.4 to 1.7 to get total national results.

Estimating TSP levels: actual 1990 levels, and levels without motor-vehicle related pollution. The WTP model derived above (equation 9-7) estimates the total annual household WTP for a change in TSP from PI to PP, where PI is the TSP level in 1990, and PP is the TSP level after all anthropogenic emissions (case I) or all motor-vehicle related emissions (case II) have been eliminated. We specify the initial pollution level, PI, to be the actual ambient air quality in each county in the U. S. in 1990, as reported by the EPA (1993d).

We then model three different TSP-reduction scenarios (i.e., three different values of PP):

I) TSP reduced from 1990 levels to the natural background levels, with no anthropogenic emissions, and

II) TSP reduced from 1990 levels to the levels that would have resulted had

- A) 10% of motor-vehicle related emissions had been eliminated, or
- B) 100% of motor-vehicle related emissions had been eliminated.

To estimate PP* and PI*, we use a simple model of emissions, dispersion, and atmospheric chemistry, developed in Report #16 and summarized in section 9.2.3 (equation 9-3). Also, as discussed in Report #13, we weight particulate emissions according to their contribution to light extinction, because TSP consists of a wide range of particulate matter (fine particles, coarse particles, sulfates, nitrates, organic aerosols, and others), which scatter and absorb light differently

Results of the analysis. The results of the analysis are summarized in Table 9-2, which shows $/kg costs, and Table 9-9, which shows total $ costs. (See section 9.2.3 for a discussion of the use of $/kg costs.)

We estimate that the cost of light extinction due to emissions attributable to motor-vehicles ranges from about $4 to $30 billion per year (Table 9-9). The uncertainty in this estimate is due in large part to uncertainty about what fraction of the total damages estimated by the hedonic model are for visibility only. The upper bound of nearly $30/billion per year seems implausible.

Table 9-2 indicates that, per kilogram of emission, direct PM and SO_X emissions have the largest visibility costs. The $/kg cost of SO_X exceeds the $/kg cost of NO_X because the fraction of SO_X that becomes particulate sulfate exceeds the fraction of NO_X that becomes particulate nitrate (and it is the nitrates and sulfates, rather than the SO_X and NO_X precursors, that reduce visibility) (see Report #16). The $/kg cost of VOCs is so small because such a small fraction of VOC emissions becomes organic aerosol.

The $/kg cost including emissions from paved and unpaved roads is much smaller than the $/kg cost of vehicular tailpipe emissions only (or tailpipe plus upstream emissions), because particulate matter from vehicles and upstream sources generally is fine, whereas most road dust PM is coarse, and the light-extinction coefficient for coarse particles is much less than the coefficient for fine particles.

9.2.5 The cost of crop losses caused by ozone air pollution from motor vehicles (Report #12)

Introduction. The detrimental effects of ambient ozone on crops, even at relatively low concentrations, are well-established. Research summarized in Report #12 suggests that ozone, either alone or in combination with nitrogen dioxide and sulfur dioxide, is responsible for virtually all U.S. crop losses resulting from air pollution. In an effort to address this problem, the Clean Air Act and its amendments include air pollution damages to vegetation as one of the criteria by which secondary national ambient air quality standards are evaluated (Adams et al., 1984).

There is, of course, an economic cost associated with this reduced productivity. In Report #12, we use a formal model of agricultural production and demand to estimate damage to eight major crops by all anthropogenic ozone air pollution, and by ozone air pollution attributable to motor-vehicle use. We use yield-loss (dose-response) functions, without a formal model of agricultural production and demand, to account for ozone damages to crops other than the eight in the formal model, and then apply a simple scaling factor to account for the minor damages by pollutants other than ozone.

Figure 9-1 demonstrates the theoretical effects on crop output of an improvement in air quality. When the air is polluted, fewer crops are produced from a given set of production inputs than when the air is clean. Thus, by reducing air pollution from existing levels (superscript *o*) to background (superscript *b*), the supply curve shifts out and probably becomes more elastic (i.e., more price responsive), from S^o to S^b. This reduces the price from P^o to P^b and increases the equilibrium quantity from Q^o to Q^b.

Society gains in economic welfare as a result of this shift in the supply curve. Consumer welfare, as measured by consumer surplus, is improved in two ways. First, the original quantity of crops Q^o is still consumed, but at the lower price P^b (areas 1 and 2 of Figure 9-1). Second, the total quantity of crops consumed is increased, resulting in a gain of new consumer surplus from the additional consumption (area 3). Producers also gain in two ways. First, improved air quality results in a lower cost of production, and saves real resource costs for the original quantity of crops (areas 2 and 4). Second, the increased production results in a gain of producer surplus from the additional revenues from the additional crops (area 5). However, producers also realize a loss in welfare due to the lower crop prices: some of the original producer surplus becomes consumer surplus as a result of the lower price (area 1).

In summary, areas 2, 3, 4, and 5 of Figure 12-1 represent the net benefit to society resulting from the shift in the supply curve. Areas 1, 2 and 3 are the net benefit to consumers; areas 4 and 5, less area 1, are the net benefit to producers.

The model. We model the net agricultural benefits of three pollution-reduction scenarios:

I) eliminate 100% of anthropogenic emissions of ozone precursors (VOCs and NO_X)

IIA) eliminate 10% of motor-vehicle related emissions of ozone precursors;

IIB) eliminate 100% of motor-vehicle related emissions of ozone precursors.[13]

A summary of the calculation procedure follows; details are provided in Report #12. In the detailed formal model, which is based on an agricultural optimization model developed by Howitt (1991), overall change in welfare as a result of a change in ozone is estimated as the sum of changes in producer surplus and consumer surplus, less changes in deficiency payments (equation 9-8). The changes in producer surplus and consumer surplus are estimated by solving a constrained surplus-maximization problem, with Howitt's model. Specifically, we solve a constrained welfare-maximization problem to find the equilibrium input resource quantities (X_{jir}) that maximize total surplus (including deficiency payments) in the crop market, subject to

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[13] We emphasize that we are modeling the benefits due to the elimination of ozone precursor (specifically, VOC and NOx emissions). Because of the nonlinearity of our simple ozone-production function (Report #16), a 10% reduction in precursor emissions does not necessarily result in a 10% reduction in ambient ozone.

the resource constraints in each region. Then we substitute these optimal X_{jir} into a production function in order to estimate the equilibrium crop production levels (Q_{ir}) in each region. Then, we substitute the Q_{ir} into a demand function in order to find the equilibrium national price for each crop P_i. We use baseline national data on prices, quantities, and demand elasticities to estimate the intercept (δ_i) and slope (β_i) of the demand curve. With estimates of X_{jir}, Q_{ir}, P_i, β_i, and δ_i, and given values for resource costs (C_{ij}), we use equation 9-9 to estimate producer surplus (including deficiency payments) and consumer surplus. We deduct deficiency payments (a federal crop price support program) in equation 9-8 because they are simply welfare transfers and do not affect net social welfare.

Formally:

$$\Delta W_{USA} = \sum_{r=1}^{12} \Delta W_r \quad \textbf{[9-8a]}$$

$$\begin{aligned} \Delta W_r &= \Delta PS_r + \Delta CS_r - \Delta DEFPMT_r \\ &= \sum_i (PS_{ir}^b - PS_{ir}^o) + (CS_{ir}^b - CS_{ir}^o) - (DEFPMT_{ir}^b - DEFPMT_{ir}^o) \end{aligned} \quad \textbf{[9-8b]}$$

where:

$$PS_{ir}^o = P_i^o Q_{ir}^o + DEFPMT_{ir}^o - MKC_{ir}^o - \sum_j HPC_{jir}^o - \sum_j VIC_{jir}^o \quad \textbf{[9-9a]}$$

$$PS_{ir}^b = P_i^b Q_{ir}^b + DEFPMT_{ir}^b - MKC_{ir}^b - \sum_j HPC_{jir}^b - \sum_j VIC_{jir}^b \quad \textbf{[9-9b]}$$

and:

$$CS_{ir}^o = \tfrac{1}{2} \cdot (\delta_i - P_i^o) \cdot Q_{ir}^o \quad \textbf{[9-9c]}$$

$$CS_{ir}^b = \tfrac{1}{2} \cdot (\delta_i - P_i^b) \cdot Q_{ir}^b \quad \textbf{[9-9d]}$$

and:

$$VIC_{jir} = X_{jir} \cdot C_{jir} \quad \textbf{[9-9e]}$$

(where the superscripts o and b have been omitted for economy of exposition)

and:

ΔW_{USA} = increase in total economic welfare (net dollar benefits) in the U.S.A. (in the markets for the eight major crops) due to a reduction in ambient ozone concentrations from 1990 levels *o* to background levels (*b* case I) or levels without 10% or 100% motor-vehicle-related ozone precursor emissions (*b* case II)

ΔW_r = increase in total economic welfare in the markets for the eight major crops) in region r due to a reduction in ambient ozone concentrations from 1990 levels *o* to background levels (*b* case I) or levels without 10% or 100% of motor-vehicle-related ozone precursor emissions (*b* case II)

ΔPS_r = increase in producer surplus, or profits, in region r due to a reduction in ambient ozone concentrations from 1990 levels o to background levels (*b* case I) or levels without 10% or 100% of motor-vehicle-related ozone precursor emissions (*b* case II)

ΔCS_r = increase in consumer surplus in region r due to a reduction in ambient ozone concentrations from 1990 levels o to background levels (*b* case I) or levels without 10% or 100% of motor-vehicle-related ozone precursor emissions (*b* case II)

PS_{ir} = producer surplus from crop i in region r

CS_{ir} = consumer surplus crop i in region r

$DEFPMT_{ir}$ = total deficiency payments for crop i in region r

MKC_{ir} = marketing costs for crop i in region r

HPC_{jir} = hedonic program cost for input j for crop i in region r

VIC_{jir} = variable input cost for input j for crop i in region r

Q_{ir} = the equilibrium quantity of crop i in region r

P_i = the equilibrium national price of crop i

C_{jir} = the constant resource cost of input j in producing crop i in region r

X_{jir} = the optimal use of input j in producing crop i in region r

δ_i = the intercept of the national demand curve for crop i with the price axis

Superscript o = "initial" ozone levels: actual levels in 1990 (estimated from data taken at ambient air-quality monitors, discussed in section 12.4.5),

Superscript b = ozone levels after either: I) all anthropogenic ozone precursor emissions is eliminated, so that ozone is reduced to the natural background level, or II) 10% or 100% of emissions of ozone-precursor pollutants from motor vehicles are eliminated (discussed below).

Subscript i = crop i (corn, cotton, wheat, barley, alfalfa, soybeans, rice, sorghum; these eight crops account for 63 percent of the total value of U.S. agricultural production)

Subscript j = input j (land, water, capital, nitrogen, and pesticides)

Subscript r = 12 agricultural regions of the United States

We model the effect of the decrease in ozone as a shift in the production function: at lower ozone levels, more output is obtained from a given set of inputs. The shift in the production function is estimated on the basis of yield-loss functions for crops.

The model of production and demand is Howitt's (1991). This model is a price-endogenous, self-calibrating, non-linear optimization program, similar in some respects to a computable general equilibrium model. In general, the model allows farmers to re-optimize their total agricultural production in response to ozone air pollution, subject to regional limits on resources, including land, water, and fertilizer, and calculates the change in consumer and producer surplus with respect to this adjusted optimum. However, the model does not allow for any technical change.

As mentioned above, the formal agricultural optimization model includes eight major crops, which in 1990 accounted for 63% of the total value of agricultural production. Because many of the crops not included in Howitt's optimization model are exposed to at least as much ozone, and are at least as sensitive to ozone, as are the eight crops included in the model, we cannot ignore ozone damages to them. To account for damages to these other crops, we scale the results of the agricultural optimization model by the ratio of total ozone damages (to all crops) estimated using simple yield-loss estimates (without a formal optimization model), to the simple yield-loss damages to the eight major crops (equation 9-10). We also apply a minor scaling factor to account for damage from pollutants other than ozone:

$$\Delta TW_{USA} = \Delta W_{USA} \cdot \left(1 + \frac{YLV_{OC}}{YLV_{8C}}\right) \cdot SFOP \qquad \textbf{[9-10]}$$

$$YLV_{OC} = \sum_{o} VP_{o} \cdot \frac{Q_{o,PP}}{Q_{o,PI}}$$

[9-11a, 9-11b]

$$YLV_{8C} = \sum_{i} VP_{i} \cdot \frac{Q_{i,PP}}{Q_{i,PI}}$$

where:

ΔTW_{USA} = total change in economic welfare in the markets for all crops due to a reduction in ambient pollutant concentrations from 1990 levels to background levels

ΔW_{USA} is as defined above

YLV = the simple yield-loss value of ozone damage to crops

VP = the value of crop production in 1990 (data from USDA Statistical Bulletins)

SFOP = scaling factor to account for damages from pollutants other than ozone (estimated to be 1.05 to 1.10, in Report #12)

Q_{PP} = yield-loss function (see equation 9-12) for background ozone levels PP

Q_{PI} = yield-loss function (see equation 9-12) for initial ozone levels PI in 1990

subscript OC = crops other than the eight included in the formal optimization model

subscript 8C = the eight crops included in the formal optimization model
subscript o = crop o not included in the formal optimization model
subscript i = crop i included in the formal optimization model

Yield-loss (dose-response) functions. A yield-loss (dose-response) function estimates the change in crop yield that results from a change in ozone concentrations. We reviewed the available literature on yield-loss functions and selected upper-bound and lower-bound functions relating levels of ozone to yields, for each of the eight major agricultural crops in the optimization model, each of the 10 most valuable crops not in the optimization model, and for the category "all remaining crops."

Typically, researchers fit experimental or econometric dose-response data to a Weibull function (Heck et al., 1984):

$$Q = \mu \cdot e^{-(\frac{OZONE}{\gamma})^{\lambda}} \quad \textbf{[9-12]}$$

where:

Q = the observed yield
OZONE = the ozone concentration in ppm (air quality data and estimates are discussed below)
μ = the hypothetical maximum yield at zero ozone
γ = the ozone concentration when Q is 0.37μ
λ = a dimensionless shape parameter

In the formal model, based on Howitt's agricultural optimization model, we use these functions to estimate yield losses of the eight major crops at the county level in the U.S. in 1990. The county-by-county yield losses then are aggregated to the regional level for the purpose of adjusting the regional production functions in the agricultural optimization model. For the purpose of scaling the results of the formal model to account for ozone damages to other crops, we apply the yield-loss functions to our estimate of national-average ozone levels, weighted by regional output and regional ozone air quality.

Air-quality modeling and data. In the formal model, we specify the initial pollution level, PI, to be the actual ambient air quality in each county in the U. S. in 1990 (EPA, 1993d). As discussed briefly in section 9.2.3 and documented fully in Report #16, we use a very simple air-quality model to estimate the pollution level after motor-vehicle related emissions are eliminated hypothetically (equation 9-3).

Note that, when we estimate the ozone level after removing motor-vehicle related emissions, we estimate the effects of a specific, "marginal" change in pollution: the difference between actual ozone (PI) and, what ozone would have been had motor-vehicle-related ozone-precursor emissions been reduced by 10% or 100% (PP). Because ozone formation is a nonlinear function of two precursor pollutants, NO_X and VOCs, the only way to model the real nonlinear effect on ozone of motor-vehicle ozone-precursor emissions is to model actual ozone levels with and without motor vehicle precursor emissions.

Results of the analysis . Table 9-3 shows the crop-damage cost per kg of emissions, and Table 9-9 shows total crop damages due to all motor-vehicle related ozone pollution.

Table 9-3 shows costs per kg of NO_X and VOC *combined* because these pollutants contribute jointly to ozone production. Note that, technically, the $/kg-[VOC+$NO_X$] results hold only for the actual proportions of VOCs and NO_X emitted in 1990. However, the results probably are reasonably accurate for up to moderate deviations from the 1990 proportions.

We estimate that pollution attributable to motor vehicle use probably causes $3 to $6 billion in agricultural damages, per year (Table 9-9) These estimated damages are much less than the damages to human health (Report #11), and thus probably constitute a relatively minor portion of the total cost of air pollution from motor vehicles.

9.2.6 The cost of material damage caused by air pollution from motor vehicles

Oxidant and acid air pollution can erode and soil a variety of man-made materials (Adams et al., 1996). Damaged materials are unsightly, and sometimes are structurally unsound.

In principle, the cost of damage to materials from motor-vehicle air pollution can be estimated by modeling the relationships between emissions and air quality, air quality and materials damage, and materials damage and economic value. Although nobody has used this approach to estimate the materials-damage cost attributable to motor-vehicle air pollution specifically, several researchers have used damage functions to estimate the cost of materials damage due to all air pollution. I review these studies of total damage, and then estimate the portion of the total attributable to motor-vehicle use[14].

Estimates. Adams et al. (1996) review 7 estimates, all done in the 1970s, of the cost of oxidant damage to materials. Total estimated damages from ranged from $1.6 to $3.8 billion (1984$). Lee et al. (1995) also review old U. S. studies of ozone damages to materials, and then estimate damage costs to materials and paints, and the cost of anti-ozonants added to rubber. With the estimates of Lee et al. (1995), Rabl and Eyre (1998) calculate total linear ozone damages in the range of £2.1 to 21 million/ppb-O_3/10^8-persons, in the U. K. Assuming population-weighted average O_3 concentrations in the U. S. in the range of 15 to 30 ppb (McCubbin and Delucchi, 1996; Table 11.7-1; Rabl and

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[14]As an alternative to our literature review and apportioning, it might be preferable to adapt the air emissions and air quality model of Report #16 to include acid deposition, update some of the published damage functions (e.g., Rowe et al., 1986; McCarthy et al., 1984), apply the damage functions to inventories of materials at risk to acid deposition (Lipfert and Daum, 1992), and develop valuation functions.

Eyre, 1998), \$1.8/£, and $2.5 \cdot 10^8$ million persons in the U. S., the cost of ozone damage to materials in the U. S. would be on the order of \$0.1 to \$3 billion -- similar to the range in Adams et al. (1996).

SO_2 and PM also damage materials, at least as much as does ozone. Crocker (1985) somewhat heroically estimates that the complete elimination of acid deposition in the eastern third of the U.S. in 1978 would save a maximum of \$2.00 billion (1978\$) in materials damages. This might imply something on the order of \$8 billion in savings for the entire U.S., in 1991 (1991\$). Adams (1986) reviews a 1985 estimate by Lareau et al. of the effect of all SO_2 and acid deposition on materials in seventeen midwestern and northeastern states. Using air-quality data, inventories of materials, and damage functions, they estimate \$2 billion in damages in 1984 dollars. This implies roughly \$5 billion in materials damage for the whole U. S., in 1991 dollars. Watson and Jaksch (1982) use demand and supply functions for "cleanliness" to estimate household benefits from reduced soiling by particulate matter. Their model and results imply that in 1991, soiling by particulate matter cost at least \$10 billion, and perhaps as much as \$30 billion, annually (1991\$).

A few studies have attempted to estimate the materials damage cost from all pollutants. One of the studies that Adams et al (1996) review estimated that materials damage from all pollutants was \$6.8 billion in 1984\$, or \$8.6 in 1991\$. Murray et al. (1985) applied materials damage functions to an inventory of materials for the South Coast Air Basin to estimate the monetary cost of increased materials maintenance caused by ambient SO_2, TSP, and ozone. The result was \$42 million in 1979\$ (\$73 million in1991\$) most of which was damage to paint. To extrapolate crudely to total national damages in 1991, I multiply by two to account for effects on omitted materials, and by 10 to 20 to scale to the rest of the country. The result is \$1-\$3 billion nationally in 1991 dollars.

Rowe et al. (1986) also review the literature on materials damages, and estimate simple damage functions for ozone (oxidation of tires), SO_2 (a wide range of effects), and TSP (household soiling). Specified for typical U.S. conditions in 1991, their damage functions estimate on the order of \$5 billion (1991\$) in damages due to ozone and SO_2, but anywhere from a few billion to a few hundred billion due to TSP. (Rowe et al. [1986] got similarly skewed results in their own application of the damage functions to California air basins.) However, I do not think that the TSP damage functions are credible.

On the basis of a brief review of two studies of materials damages in Southern California (one of which is the Rowe et al. [1986] study mentioned above), the Congressional Research Service (Behrens et al., 1992) estimates that total air pollution damages to materials from motor vehicles nationwide cost at least \$300 million (year of dollars unspecified).

The literature reviewed above suggests that total damages to materials, including soiling, from all sources of pollution, are between \$5 and \$30 billion, in 1991\$. However, I am skeptical of the higher values, and believe that \$20 billion is a more reasonable upper bound. The motor-vehicle contribution might reasonably be estimated as lying between 20% to 40%[15]. Thus, as indicated in Table 9-9, I assume that damage to materials from motor-vehicle pollution cost $ 1.0 billion to $8.0 billion in 1991$[16].

9.2.7 The cost of forest damage caused by air pollution from motor vehicles

Ozone and acid air pollution can injure trees (Kolb et al., 1997; Adams et al., 1996; Taylor et al., 1994). Although it is difficult to isolate the causes of forest decline, there is reasonably compelling evidence that air pollution -- especially ozone and nitrogen deposition -- is at least partially responsible for some of the damages (Taylor et al., 1994; McLaughlin, 1985)[17]. Experiments in which seedlings, mature trees, or branches are enclosed in open-top, open-bottom, or "branch-exposure" chambers, and administered controlled concentrations of ozone, clearly demonstrate a wide range of detrimental effects, including reduced net photosynthesis, reduced root growth, and visible damage to leaves (Kolb et al., 1997). At the macro scale, there appears to be a link between ozone pollution and the decline of Jeffrey, ponderosa, and eastern white pines.

In general, damaged forests produce less timber than do healthy forests, and are less appealing as recreational sites and less suitable for wildlife habitat. Ideally, we would estimate the damages by modeling the relationships between emissions and air quality, air quality and forest damage, and forest damage and economic value. However, we could not find any pollutant-damage functions or valuation functions that could be applied to national data. This is not surprising: not only it is difficult to isolate the anthropogenic from the natural causes of forest decline, it is difficult to sort out the effects of different pollution burdens (ambient ozone, acid deposition directly on trees, acid deposition on soil, the combined effects of ozone and acid deposition, and excessive nitrogen deposition) (Taylor et al., 1994; Winner, 1994; MacKenzie and El-Ashry, 1988; McLaughlin, 1985)

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[15]It is possible to model the motor-vehicle contribution, but given the enormous uncertainty in the estimates of the total, the modeling is not worth the effort.

[16]The relationship between the external cost of materials damage and the external cost of crop damage from emissions from electricity generation imply a higher range: a detailed study of the costs of air-pollution from electricity generation in Europe finds that damages to materials are an order of magnitude higher than damages to crops (Krewitt et al., 1996), and the New York Environmental Externalities Model estimates that the damages to materials from power-plant pollution are at least twice the damages to crops (Rowe et al., 1995). If these relationships apply to air pollution from motor vehicles in the U.S., then damages to materials from motor-vehicle air pollution in the U.S. are on the order of $5-30 billion annually. On the other hand, Rabl (1999) finds that air pollution damages to buildings in France are two orders of magnitude less than damages to humanhealth.

[17]Volume 4, Number 4 (1994) of *Ecological Applications* is devoted to air pollution and terrestrial ecosystems. Volume 98, Number 3 (1998) of *Environmental Pollution* covers the effects of air pollution on forests in central and astern Europe.

Instead, I base my estimates on a review of the literature. Of course, given the difficulties sketched out above, the estimates in the literature are little more than illustrative. As Adams (1986) notes, "because of the uncertain state of biological science few attempts have been made to quantify the possible economic consequences," and the few available estimates of the dollar damages "at best, are preliminary and primarily of qualitative value" (p. 467). More recently, Adams et al. (1996) suggest that "conclusions regarding effects on forests and ecosystems must await the acquisition of additional data and possible refinements in ecological and economic methods" (p. 5-238).

Estimates. Adams (1986), Behrens et al. (1992), and Adams et al. (1996) cite an estimate, done for the National Acid Precipitation Assessment Program, that acid rain causes $0.34 to $0.71 billion (1984 dollars) worth of damage annually to hardwood and softwood forests in the eastern U. S. The estimate was based on spatial equilibrium model of stumpage and forest products, and assumed that acid rain reduced growth by 10% to 20%.

Crocker (1985) estimates that the complete elimination of acid deposition in the eastern third of the U.S. in 1978 would save a maximum of $1.75 billion/year in forest damages, in 1978 dollars. This might imply something on the order of $4 billion for all forests in the U.S, in 1991 dollars. Adams (1986) and Adams et al. (1996) cite Crocker's (1985) estimate, but describe it as "naive," because it assumes simply that acid deposition causes a 5% reduction in products, and then values the reduced output at average prices. Adams (1986) and Adams et al. (1996) also cite a similar estimate by Crocker and Forester that acid deposition caused $1.5 billion (1981 Canadian dollars) worth of damages to Canadian forests in 1981.

Peterson et al. (1989) review several studies of the value of various aspects of forest quality, including watershed protection, recreation, habitat preservation, and ecosystem functions. A survey of recreators and homeowners in the Los Angeles region found that respondents were willing to pay $130 (recreators) or $225 (property owners) per household per year for management efforts to offset ozone damage to all forests throughout the U.S. In the same study, the total cost of ozone damage to the San Bernadino and Angeles National Forests alone was estimated to be $27 billion to $144 million per year. (The Congressional Research Service study, mentioned above, reviewed these results.) These results suggest that pollution damages to forests in the U.S. cost as much as a few billion per year.

Haynes and Adams (cited in Adams et al. [1996]) assume that acid deposition causes losses of 6% to 21% in eastern softwood forests, and then apply an econometric model to estimate that these losses are worth $1.5 to $7.2 billion (1986 dollars) per year.

A detailed study of the costs of air-pollution from electricity generation in Europe estimates that damages to forests are not more than half of the damages to crops (Krewitt et al., 1996). If this relationship applies to air pollution from motor vehicles in the U.S., then damages to forests from motor-vehicle air pollution on the order of $1 billion annually.

The literature reviewed above suggests that total pollution damages to forests are in the range of $0.5 billion to perhaps as much as $5 billion. The motor-vehicle contribution might reasonably be estimated as lying between 30% to 40%. Thus, as indicated in Table 9-9, I assume that motor-vehicle damage to forests cost $0.2 billion to $2.0 billion in 1991.

This estimate does not include pollution damages to ecosystems other than forests. (See Adams et al. [1996] for brief discussion of the effect of pollution on ecosystems other than forests.)

9.2.8 Climate change damage costs due to lifecycle CO_2-equivalent emissions of greenhouse gases from motor vehicles

Most atmospheric scientists believe that an increase in the concentration of so-called "greenhouse gases" --- primarily carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), ozone (O_3), chlorofluorocarbons (CFCs) and aerosols (particulate matter, or PM) -- will increase the mean global temperature of the earth. Recently, an international team of scientists, working as the Intergovernmental Panel on Climate Change (IPCC), has concluded that "the balance of evidence suggests that there is a discernible human influence on global climate" (IPCC, 1995a, p. 5). In the long run, this global climate change might affect agriculture, coastal developments, urban infrastructure, human health, and other aspects of life on earth (IPCC, 1995b).

Highway vehicles are a major source of the greenhouse gases thought to be responsible for global warming. In the U.S., for example, motor vehicles account for as much as 30% of total CO_2 emissions from the use of all fossil fuels (DeLuchi, 1991).

In principle, the global-warming "damage" cost to the U. S. of greenhouse-gas emissions from motor vehicles in the U. S. can be estimated with the following simple expression:

$$TCGHG = CTCO2 \cdot \sum_{V} LCCE_V \cdot M_V \qquad \textbf{[9-13]}$$

where:

TCGHG = the total cost to the U.S. of emissions of greenhouse gases (GHGs) attributable to the use of U.S. motor vehicles ($) (calculated results shown in Table 9-4)

CTCO2 = the marginal damage cost to the U.S. of CO_2-equivalent emissions in the U.S. ($/g) (Table 9-4; based on estimates discussed below and summarized in Table 9-5)

$LCCE_V$ = lifecycle CO_2-equivalent emissions of GHGs per mile of travel by motor-vehicle class *V* (g/mi) (Table 9-4; discussed below)

M_V = total annual travel by vehicle class *V* in 1990 (miles) (Table 9-4)

subscript *V* = motor vehicle classes: light-duty passenger vehicles (including motorcycles and buses), light-duty trucks, heavy-duty trucks

In the following paragraphs, I discuss the emissions (LCCE) and the cost (CTCO2) input parameters.

Lifecycle CO_2-equivalent emissions of greenhouse gases. To estimate emissions of GHGs from motor vehicles, I ran the Lifecycle Emissions Model (LEM) developed by Delucchi (2003). The LEM estimates CO_2-equivalent emissions from the lifecycle of light-duty and heavy-duty gasoline and diesel vehicles. In the LEM the lifecycle includes: the recovery and transport of primary energy feedstocks, the production of fuels from feedstocks, the distribution of fuels to end users, the end use of fuels in vehicles, the servicing and maintenance of transport modes, the building of major energy facilities (in the cases where the emissions were likely to be important), the manufacture of materials for motor vehicles, and the assembly of motor vehicles.

The LEM includes emissions of all of "direct" and "indirect" GHGs: CO_2, CH_4, N_2O, CFCs, non-methane hydrocarbons (NMHCs), carbon monoxide (CO), nitrogen oxides (NO_X), sulfur oxides (SO_x), particulate matter (PM), hydrogen (H_2), and other gases. Emissions other than CO_2 are converted to the amount of CO_2 with an equivalent impact and then added to actual CO_2 emissions to produce a "CO_2-equivalent" total. (See Appendix D of Delucchi [2003] for details on CO_2-equivalency calculations.)

The LEM can be run for any target year from 1970 to 2050, for up to 30 different countries. Given a target year and designated country, the LEM internally selects or estimates target-year and designated-country values for all important parameters in the lifecycle calculation (fuel characteristics, energy-efficiency parameters, emission factors, motor-vehicle characteristics, material flows, land use, etc.) *except* motor-vehicle fuel economy, which is an independent user-input parameter. For this analysis, I set the LEM to simulate the U. S. in 1990, and then independently specified fuel-economy levels for each class of motor-vehicle in 1990. (Fuel economy for each class was calculated as actual 1990 VMT divided by actual 1990 fuel use by vehicle class [Table 10-3 of Report #10 in the UCD Social Cost Series].) Given this, the LEM produced estimates of total lifecycle CO_2-equivalent emissions by vehicle class for the U. S. in 1990. These LEM output estimates were used as the basis of the ranges estimated in Table 9-4.

Implicit in this calculation of lifecycle emissions of GHGs are two assumptions: first, that a change of X gallons of demand for fuel F causes a change of X gallons of refinery output of F and a change in production of crude oil equal to the amount required to produce X gallons of F; and second, that emissions from U.S. producers and refiners are representative of the emissions from all of the producers and refiners affected by changes in U.S. transportation demand. Neither assumption is strictly correct, because price changes affect petroleum demand in nontransportation sectors, and because oil, fuels, and vehicles are produced and traded in a world market. I suspect but do not demonstrate that the error introduced by failing to account for the effect on prices and consumption may not be small. small. On the other hand, I believe that the second assumption is reasonable, because a change in U.S. demand likely will affect U.S. refiners mainly, and because in any case the energy intensity and emissions of oil production and refining in other countries is similar to that in the U.S. (Also, recall that in the case of global warming, the location of the emissions does not matter much.)

The damage cost in the U. S. of CO_2 emissions in the U. S. In this section, I first review estimates of the *global* cost of CO_2 emissions per ton. Then I estimate the fraction of global damages that the U. S bears.

Estimates of the actual damages. The global damage cost per unit of CO_2 or CO_2-equivalent emitted depends on the level of emissions, the response of the climate, the scope and magnitude of damages considered, the time horizon, the discount rate, the normative treatment of risk and low-probability/high-cost events, and other factors. None of these are easy to model, and as a result, estimates of the damage cost, summarized in Table 9-5 and reviewed below, vary widely. (The OECD [1991] and Pearce [2003] review some of the difficulties and uncertainties in modeling the economic impacts of climate change.)

1). Cline (1992). Cline's (1992) is one of the first detailed analyses of damage costs per se. Cline analyzes the effects of global warming on agriculture, forests, species, sea level, space heating and cooling, personal comfort, human health, human migration, hurricanes, construction, leisure activities, water supply, urban infrastructure, and air pollution. He estimates that a doubling of the concentration of CO_2-equivalent gases, and a warming of 2.5^{o} C, would cause economic losses of 1 to 2% of world or U.S. Gross Domestic Product (GDP), and that a long-term warming of 10^{o} C would cause losses of 6 to 12% of world or U.S. GDP. Cline (1992) assumes a discount rate of 1.5% per year. His quantitative damage estimate does not include impacts on human morbidity (he does estimate the impact on mortality), the construction industry (he notes that the effects could be positive), personal comfort, and urban air pollution other than tropospheric ozone. It probably also underestimates the cost of species loss, the cost of agricultural losses, and the cost of human mortality due to global warming[18]. Of the costs that are estimated, the largest are agricultural losses (net of an allowance for the benefit of carbon fertilization due to elevated CO_2 levels) , increased demand for electricity for space cooling (net of reduced costs for space heating), stress on water supply, consequences of sea-level rise, human mortality, and species loss.

Cline (1992) does not express his damage estimates per ton of CO_2; rather, he defines an aggressive CO_2-abatement strategy, and then determines if the benefits of this strategy exceed the costs, under various scenarios. However, my analysis of his results, and some of his statements, indicate that his estimates translate into damages of $4 to $40/ton-C-equivalent (1991$; Cline actually presents costs in 1990$) in the short run, and much higher in the long run. Consistent with this, Hohmeyer (1996) shows that Cline estimates costs of $5.8 to 221/ton-C-equivalent (in 1990$).

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[18]Cline (1992) estimates the mortality cost to be $5.8 billion: an assumed 9,800 deaths annually due to global warming multiplied by $595,000 per life. However, the $595,000/life includes only the value of foregone earnings due to premature death. Cline recognizes that this "human capital" approach to valuing life ignores the valuable part of humanity that is unrelated to earning a wage, and that the more theoretically sound estimates of the statistical value generally exceed $2 million. If one uses $2.5 million/life, the resulting total mortality cost is $25 billion per year.

2). Nordhaus (1993) uses a dynamic integrated climate-economy (DICE) model to estimate the costs of global warming. The DICE model uses the following damage function:

$$d(t) = 0.0133 \cdot \left(\frac{T(t)}{3}\right)^2 \cdot Q(t) \qquad \textbf{[9-14]}$$

where:

d(t) = the loss of global output from greenhouse warming at time t
T(t) = the global temperature at time t, relative to the pre-industrial temperature
Q(t) = world output at time t

Equation 9-14 estimates that the damages from a 3^{o} C temperature increase would be 1.33% of world output.

Nordhaus uses the DICE model to estimate the net benefits of several greenhouse-gas policies. One is called the "optimal" policy, in which "the nations of the world gather to set the efficient policy for internalizing the greenhouse externality" (p. 39). Nordhaus represents this efficient internalization with a carbon tax. In the optimal policy, the carbon tax rises from $5/ton-C-equivalent (1989$) in 1990 to $20/ton-C-equivalent in the year 2000, and continues rising thereafter. (The "equivalent" in "per ton-C-equivalent" means "per ton of carbon or anything else with the equivalent effect on global climate".) Because the optimal tax should be equal to the present value of marginal damages, one can infer that the DICE model estimates damages of $5 to $20/ton-C-equivalent (in 1989$; I use the U. S. GDP implicit price deflators to convert to 1991$/ton-C in Table 9-5).

In earlier work, Nordhaus (1991) estimates that a doubling of CO_2-equivalent concentration and a resultant temperature increase of 3^{o}C would cause damage to farms, forests, fisheries, construction, water transportation, energy and utilities, real estate, and hotels and recreation, equal to about 0.25% of U. S. national income. He then speculates that if the unquantified damages in other sectors of the economy were added, the total damage would be 1.0% or perhaps 2.0%, but probably not more than 2.0%, of total output. He converts these percentages into 1989$/ton-C as a function of the difference between the real interest rate on goods (r) and the growth rate of the economy (h):

r - h % per year	*Present value of climate damages, 1989$-ton-C-equivalent, for damages as a percentage of world output*		
	2.0%	*1.0%*	*0.25%*
0	65.94	32.97	8.24
1	14.65	7.33	1.83
4	2.44	1.22	0.31

I use the U. S. GDP implicit price deflators to convert the high and the low to 1991$/ton-C in Table 9-5. As above, the "equivalent" in "per ton-C-equivalent" means "per ton of carbon or anything else with the equivalent effect on global climate".

Many of Nordhaus' damage estimates come from a 1989 report by the EPA. In 1991, EPA published a revised and corrected version of the 1989 report. According to Morgenstern (1991), if Nordhaus had available the revised and corrected EPA figures, his lower-bound estimate of damages of 0.25% of output would have doubled to 0.50%.

Parry (1993) extended Nordhaus' (1991) work to estimate the "insurance" value of reducing the potential *variability* in future consumption due to warming, and found that it is not worth paying now to reduce future uncertainty in global warming, unless the risks are unimaginably large, or the discount rate is near zero.

3). Ayres and Walter (1991) re-examine early estimates by Nordhaus (papers published in 1989 and 1990; not referenced here) of the damages from a rise in sea level due to global warming, and conclude that the seal-level damages (which according to Ayres and Walter constituted 92% of the damages estimated by Nordhaus in his 1989 and 1990 papers) could amount to at least 2.1 to 2.4% of gross world income, or $30-$35/tonne-CO_2, apparently in 1981$. Although they express the results per tonne of CO_2-equivalent, they probably mean per tonne of carbon in CO_2-equivalent, because Nordhaus' original estimates, which they revise, probably are in $/ton-C-equivalent. (In Nordhaus 1991 and 1993 papers, the results are given in $/ton-C-equivalent.) Therefore, I assume that Ayres and Walters (1991) estimate $30-$35/tonne-C, in 1981$, and use the U. S. GDP implicit price deflators to convert to 1991$ in Table 9-5.

4). Fankhauser (1994) presents the global damage cost per ton of carbon as a probability distribution, assuming that the pure rate of time preference, the income elasticity of marginal utility or rate of risk aversion, future emissions, the shape of the damage function, and all other key parameters are random variables with triangular distributions. In the case of the discount rate and rate of risk version, the points of the triangular distribution are: p: 0.0% - 0.5% [peak of distribution] - 3.0%; and w: 0.5 - 1.0 [peak] - 1.5. Fankhauser follows Cline's (1992) treatment of discounting, and uses data from Nordhaus to estimate some parameter values in his model. His damage function is calibrated with respect to baseline estimates of the % loss in world GNP due to a doubling of CO_2 levels. His $/ton-C damage estimates are[19]:

[19]Fankhauser (1994) cites another study that also estimates $10 to $20/ton-C.

	Period 1991-2000	*Period 2021-2030*
5th percentile	6.2	9.2
mean	20.3	27.8
95th percentile	45.2	64.2

Fankhauser (1994) shows that even higher costs result if one assumes that the parameter that determines the shape of the damage function and the parameter that determines the climate sensitivity are distributed log-normally, with an open upper bound, rather than triangularly. The log-normal distribution allows for low-probability, catastrophic events.

I assume that his estimates are in 1990$, and use the U. S. GDP implicit price deflators to convert to 1991$ in Table 9-5.

5). Titus (1992) estimates the economic cost of the effect of global warming (a doubling of CO_2 levels, and an equilibrium temperature increase of 3.7^o C) on agriculture, energy consumption, sea level, health, automobile air conditioners, ozone air pollution, water resources, and forests in the U.S. He presents the present value of marginal damages to the U. S. per gallon of gasoline burned (1990$/gallon), for three different discount rates, and with two different warming models: the Princeton Geophysical Fluid Dynamics Laboratory (GFDL) model, and the Goddard Institute for Space Studies (GISS) model:

	Discount rate		
	2%	*3%*	*4%*
GISS model			
before 2100	0.287	0.165	0.103
in the long run	0.970	0.247	0.120
GFDL model			
before 2100	0.426	0.252	0.140
in the long run	1.356	0.365	0.186

I multiply by 375 gallons-gasoline/ton-C-gasoline (DeLuchi, 1991) to convert these to $/ton-C, and use the U. S. GDP implicit price deflators to convert to 1991$ in Table 9-5.

The values shown in parentheses correspond to the upper bound before 2100. This can be compared with Nordhaus' estimate of the $/ton-C damage cost in the year 2100.

I note, though, that I cannot reconcile Titus' results expressed per gallon (above) with his estimate that a reduction of 10 billion metric tons of carbon emissions would create annual benefits of $1-2 billion during much of the next century. As shown in Table 9-5, the cost-per-gallon estimates translate to on the order of $100/ton-C, whereas the present value of $1-2 billion/year over 100 years divided by 10 billion metric tonnes implies less than $10/ton-C.

6). Pearce et al. (1992) estimate damages of 1% to 3% of Gross World Product in 2050, with associated costs of $10 to $30/metric-tonne-C. I assume that the original estimates are in 1990$, and use the U. S. GDP implicit price deflators to convert to 1991$ in Table 9-5.

7). Hohmeyer (1996) cites a 1992 estimate by him and Gärtner of the damages from a doubling of CO_2-equivalent concentration:

Cumulative damages through the year 2035	*(10^{12} US $)*
1. Temperature rise	1.9
2. Sea level rise	2.9
3. Precipitation changes	901
4. Storms	1.5
5. Total	907.3
share of total to CO_2 (60%)	544.0
damages in $/ton-$CO_2$	*220*

In the Hohmeyer and Gärtner work, essentially all of the damages result from changes in precipitation. Hohmeyer (1996) elaborates: "The main effect resulted from changes in agricultural production due to different changed precipitation patters, higher evaporation, higher frequency of droughts, and decreased availability of water for irrigation purposes. The main effect caused by a drop in agricultural production was the resulting starvation of hundreds of millions of people in the poorest countries of the world" (p. 71). I infer from this that the bulk of the $900 trillion in damages estimated by Hohmeyer and Gärtner is the value of the lost life: perhaps something like 400 million people x 2 million dollars per life = 800 trillion dollars. This nicely illustrates the importance of assumptions regarding changes in human mortality due to changes in global climate.

8). The California Energy Commission (CEC, 1992) has adopted a damage estimate of $28/ton-C (1989$), although CEC staff recommended a value of $41/ton-C ($11.18/ton-CO_2) based on national estimates of the cost of reducing emissions of carbon dioxide. I use the U. S. GDP implicit price deflators to convert to 1991$ in Table 9-5.

9). Montgomery (1991) states that economic benefits "associated with ameliorating global climate change..could amount to no more than 0.5% of GNP" (p. 1), but he does not elaborate.

10). Tol (1995) estimates that a doubling of CO_2 equivalent concentrations relative to pre-industrial levels, with a concomitant average warming of 2.5 °C, would cause damages of 1.9% of global GDP, or 315.7 billion $1988.

Tol (1995) estimates damage costs in nine global regions, and aggregates the regional costs to a global total. He uses damage cost functions with linear and quadratic components for several loss categories, with basic assumptions as follows:

Loss Category	*Tangible Damages*	*Intangible Damages*	*Relative to parameter*
coastal defense	0.5 ME/LI 0.5 RE/LI	none	sea level
dry land loss	1.0 ME/LI	none	sea level
wetland loss	0.25 ME/LI 0.25 RE/LI	0.25 ME/LI 0.25 RE/LI	sea level
species loss	none	0.5 ME/QU 0.5 RE/QU	temperature
agriculture	0.75 RE/LI 0.25 RE/QU	none	temperature
amenity	none	0.17 ME/QU 0.83 RE/QU	temperature
life/morbidity	none	0.17 ME/QU 0.83 RE/QU	temperature/ hurricane
emigration	1.0 ME/LI	none	sea level
immigration	none	1.0 ME/LI	sea level
natural hazards	0.75 ME/LI 0.25 ME/QU	none	hurricane

ME = depends on magnitude of effect of given parameter; RE = depends on rate of effect of given parameter; LI = linear; QU = quadratic

He also assumes that a statistical life is assumed to be worth $250,000 plus 175 times the average income per capita (thus ranging from $299,000 to $3.4 million).

11). More recently, Tol (1999) has used a long-run, dynamic model of climate change and economic impact, called "FUND," to estimate the marginal cost of emissions of CO_2, CH_4, and N_2O. His model includes:

i) nine world regions (OECD-American, OECD-Europe, other Europe and the former Soviet Union, Middle East, Latin America, South East Asia, Centrally Planned Asia, and Africa);

ii) three greenhouse gases (CO_2, in a five-box model, and CH_4 and N_2O, represented by geometric decay functions);

iii) exogenous scenario parameters for economic growth, population, urban population, autonomous energy efficiency improvements, decarbonization of energy use, and CH_4 and N_2O emissions;

iv) impacts on sea level rise, species, agriculture, extreme weather (such storms, floods, hurricanes), heat stress, cold stress, and malaria

v) valuation of the tangible and intangible effects of the impacts of climate change.

The model runs from 1950 to 2200 in one-year steps. The scenario variables are slightly "perturbed" by the climatic effects: for example, mortality caused by climate change affects the world population tracked in the model.

Using the baseline IPCC scenarios, Tol's (1999) model calculates the following marginal damage costs per metric tonne of carbon in CO_2 emitted between 1995 and 2004, as a function of the discount rate, and other parameters ($/t-C [$/t-CO_2]; in 1990 $):

discount rate -->	1%	3%	5%
no equity weights, base case	73 [20]	23 [6.3]	9 [2.5]
equity weights, best guess	171 [47]	60 [16]	26 [7.1]
equity weights, 5th percentile	81 [22]	26 [7.1]	11 [3.0]
equity weights, 95% percentile	512 [140]	178 [49]	77 [21]

In the "no equity weights" cases, Tol (1999) assumes that utility is a linear function of monetary income, and hence that such things as the value of mortality vary from region to region in proportion to the per-capita income, with the result that damages in poor countries are relatively small. In the "equity weights" cases, Tol (1999) assumes that the regional welfare is the natural log of per-capita income, with the result that differences in per-capita welfare losses (e.g., due to mortality) are much less than differences in per-capita income. The equity weights matter a great deal (note the nearly 3-fold difference between the "no equity weights" and "equity weights" cases) because the vast majority of the damages occur in relatively poor regions: Southeast Asia, Latin America, Middle East, and Africa.

Roughly 90% of the damages are due to sea level rise and extreme weather. In a Monte-Carlo simulation, Tol (1999) finds that the distribution of damages is skewed towards the right (large damages)

12). Goodland and El Serafy (1998) review the literature, including many of the studies reviewed here, and recommend a value of $6.80/tonne-$CO_2$ ($25/tonne-C; year of $ not specified).

13). Tol (2003a) applies an updated version of his "FUND" model (FUND2.0) to estimate marginal damage costs per metric tonne of carbon as a function of the pure rate of time preference (TP, %/year), the use of equity weighting (EW) or not (no EW), and the time horizon (TH) ($/tonne-C; $/tonne-$CO_2$ are shown in parentheses) (apparently year-2000 $):

	TP = 0%		TP = 3%	
	no EW	EW	no EW	EW
TH = 2050	3.2 (0.9)	3.2 (0.9)	1.8 (0.5)	1.4 (0.4)
TH = 2150	11.4 (3.1)	24.9 (6.8)	2.2 (0.6)	2.4 (0.7)

Equity weighting places greater value on a dollar of damages in a poor country than on a dollar of damage in a rich country; thus, because damages occur disproportionately in poor countries, the use of equity weighting increases estimated marginal damages. Because damages tend to occur far into the future, the use of a lower discount rate results in higher marginal damage costs. Note that the costs estimated by this updated version of FUND are much lower than those reported in Tol (1999) (item 11 here).

14). Tol (2003b) reviews 88 estimates of the marginal costs of CO_2 emissions, from 22 different studies. Qualitatively, he finds that market impacts are lower than initially thought, and in some cases are positive; and that developing countries are more vulnerable to climate change than are developed countries
Quantitatively, he combines the estimates into a probability density function, and finds that it is skewed strongly to the right (i.e., with a long "tail" of high-cost estimates): the mode is \$5/tonne-C, the median is \$13/tonne-C, and the mean is \$104/tonne-C. If only peer-reviewed studies are included, the highest estimates are omitted, and the mean is cut in half to \$57/tonne-C. Studies with equity weighting or lower discount rates produce higher marginal damage estimates (see also Pearce [2003]). Tol (2003b) concludes that the "best guess for the marginal costs of carbon dioxide emissions is \$5/tC," and that "it is unlikely that the marginal costs of carbon dioxide emissions exceed \$50/tC and are likely to be substantially smaller than that". His preferred range therefore appears to be about \$5 to \$50/tonne-C, or about \$1 to \$14/tonne-CO_2, presumably in year 2000 dollars.

15). Pearce (2003) provides an excellent current review and analysis of estimates of the marginal social damage cost of emissions of CO_2. His review includes Nordhaus (1991), Fankhauser (1994), Tol (1999, 2002a, 2002b), and others not covered in this report. His analysis highlights several key factors in models of the marginal social damage cost of CO_2 emissions:

Adaptation: Early models generally ignored the possibility that people will adapt to climate change and thereby mitigate damages, albeit at some "adaptation" cost. More recent models that incorporate adaptation produce much lower estimates of damages than do models without, all else more-or-less equal.

Catastrophic effects: Some models include a small probability of catastrophic effects, which of course tends to increase marginal damages.

Benefits: On the other hand, some models overlook the possibility of *benefits* of climate change, which by definition tend to offset the estimated marginal costs.

Equity weighting: To the extent that net damages tend to fall disproportionately on the poor, models that assign a greater "weight" to $1 damage to poor people than to $1 damage to rich people will produce greater (weighted) damage estimates than will models that don't assign such equity weights.

The discount rate: Because damages from CO_2 emitted today occur in the distant future, the marginal damage cost of a current emission, which is the present value of future damage streams, depends greatly on the discount rate. Theoretical arguments indicate that for long-term problems such as climate change the discount rate should actually decline with time (Weitzman, 1998). Pearce (2003) shows that adopting a discount rate that declines with time will reduce estimates of marginal damages.

Based on this review and analysis of the literature, Pearce (2003) suggests the following best estimates of the marginal social damage costs of CO_2 emissions (year-2000 $-metric-tonne/C ($-t/CO_2))

Base case: based on models that incorporate adaptive behavior and (in some cases) catastrophic events but also benefits of climate change, with *no* equity weighting and a constant discount rate:	4 to 9 (1 to 3)
Same as base case, but with equity weighting	4 to 23 (1 to 6)
Same as base case, but with equity weighting & time-varying discount rate	6 to 41 (2 to 11)

In 1991 $, these estimates would be 20-30% lower.

<u>*CO_2 "adders," and the cost of control.*</u> It is interesting at least to compare estimates of actual damages with estimates of the cost of "controlling" CO_2 emissions. Although a cost-of-control estimate per se is not the same as a damage estimate, some analysts do -- in some cases legitimately -- use the former as a guide to estimating the latter (see the discussion in section 9.1.4). As mentioned above, the staff of the California Energy Commission (CEC) has recommended a damage-value of $11.18/ton-$CO_2$ ($41/ton-carbon), based on national estimates of the cost of reducing emissions of carbon dioxide. The CEC also notes that other states have adopted estimates ranging from $1 to $24/ton-$CO_2$. Harrison and Nichols (1996) show that the CO_2 "adders" adopted by public utility commissions in seven states range from a low of $1.10/ton-$CO_2$ in New York (1989$) to a high of $40.00/ton-$CO_2$ in Oregon (1990$), with an average of about $17/ton-$CO_2$. Most of the estimates are based on the cost of planting trees. Finally, the National Academy of Sciences (NAS) (1991) characterizes costs of control between $1 and $9/ton-$CO_2$-equivalent as "low," and costs between $10 to $99/ton as "moderate". At something of a stretch, one might infer that the NAS' qualifiers "low" and "moderate" are with respect to the actual damages, and hence that the NAS believes that the damages are in the range of, say, $10 to $100/ton-$CO_2$.

Carbon taxes. It also is interesting to note, as yet another point of reference, that Denmark, Finland, the Netherlands, Norway, and Sweden have adopted carbon or energy taxes that range from $4 to $40/ton-$CO_2$ (1994$), with an average of around $16/ton-$CO_2$ (Muller, 1996). Again, at a bit of stretch, one might assume that these taxes represent political willingness-to-pay to reduce GHG emissions[20].

Summary, and my estimate. The literature review above may be summarized as follows ($/ton-$CO_2$ [or $/tonne-$CO_2$; the difference is not significant here], with central tendencies or best estimates parentheses)

Damage estimates, Table 9-5: Cline (short run), Nordhaus, Ayers and Walters, Fankhauser, Pearce (literature review and analysis), Montgomery, CEC, Tol (most recent work and literature reviews), Goodland and El Serafy	1 - 20 (best models: 1 – 10)
High damage estimates, Table 9-5: Cline (long run), Titus, Hohmeyer, Tol (1999, upper ends, discount rate 1%)	60 - 200
CO_2 adders and carbon taxes	1-40 (10-20)

Most of the damage estimates, carbon taxes, and CO_2 adders *tend*, very roughly, towards $1 to $20/ton-$CO_2$. However, the most recent best estimates of marginal damages, as presented in Pearce (2003) and Tol (2003a, 2003b), are in the lower half of this range, or $1 to $10/ton-$CO_2$.

Of course, damages and control costs can vary much more widely than this. As indicated in the review above, some current models estimate overall net *benefits* of CO_2 emissions, while others estimate net costs on the order of $100/ton-$CO_2$ or higher. Indeed, as Hohmeyer (1996) correctly points out, if one compounds the uncertainty in key parameters, such as the discount rate, the number of lives lost due to global warming, and the value of those lives, the overall uncertainty can span several orders of magnitude. In a similar vein, Krupnick (1993) observes that climate-change damages can range from "zero to catastrophic." Such a wide range makes it difficult to choose low and high $/ton values for the cost analysis.

Still, the analytical situation is not quite so dire, because there are reasons to question the high estimates of damages (Pearce, 2003; Tol, 2003b). For example, of the four high damage estimates presented above, at least two are suspect. As noted above, the high estimate of Titus (1992) might actually be a mistake. And the estimate of

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[20]However, one certainly would not want to go so far as to assume that each country set the tax at what it believes to be the marginal damages from CO_2 emissions. Although the taxes were designed to contribute to national greenhouse-gas reduction goals, some of them partly replaced existing taxes in the energy base, and some appear to be directed as much towards energy consumption in general as to CO_2 emissions in particular (Muller, 1996).

Hohmeyer and Gärtner (in Hohmeyer, 1996) is based on the perhaps dubious assumption that hundreds of millions will die from famine caused by global warming. Even if one accepts that global warming will wreak havoc with precipitation in the regions of the world least prepared to adapt (and this in itself is questionable, given the limited ability of current models to predict changes at the regional scale), it still does not follow that massive famine will result, because food production and distribution is determined by a host of social and political conditions as well. For example, as regards the agricultural sector, the Economic Research Service (1996) points out that:

> A collection of recent research efforts at the farm, national, and global levels finds that while climate change can have impacts on the agricultural sector, there is considerably more sectoral flexibility and adaptation potential for U.S. agriculture than was found in earlier analyses. Negative effects of climate change on agriculture found in earlier studies were overestimated because they did not account for economic adjustments and adaptation, nor did they consider the broader economic and environmental implications of social changes that are likely to occur in the extended time frame of climate change.

In general, the estimates of very high damages probably do not account for all of the ways in which people will adapt to climate change and thereby minimize the damages therefrom.As a result, these high estimates are unreliable (Pearce, 2003). On the other hand, some of the models that estimate net *benefits* from climate change may not properly account for the possibility, however small, of extremely large, almost catastrophic damages[21].

With these considerations, one perhaps can narrow the "reasonable" range of damages to a single order of magnitude, \$1 to \$10/ton-CO_2.

The fraction of global damages that the U. S. bears. Most of the studies in Table 9-5 estimate global damages. Because we have restricted our analysis of the social cost of motor-vehicle use to the U.S., we must determine the fraction of global \$/ton-C damages that the U. S. bears.

In many models, damages are estimated or expressed a fraction of world output (Table 9-5). With this consideration, we estimate \$/ton damages in the U. S., given \$/ton damages in the world, as follows:

$$DT_{US} = DT_W \cdot \frac{D_{US}}{D_W} = DT_W \cdot \frac{\frac{D_{US}}{GDP_{US}}}{\frac{D_W}{GDP_W}} \cdot \frac{GDP_{US}}{GDP_W} \qquad \textbf{[9-15]}$$

where:

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[21] In this respect, Duong (1998) shows that the cost of being unprepared for extreme, unanticipated climate change generally is greater than the cost of investing in mitigation more than turns out to have been necessary, and, correspondingly, that "there is a large benefit in purchasing insurance against climate change by early action to mitigate greenhouse gas emissions" (p. 61).

DT = dollar damages from climate change, in the U. S. or in the world, per ton of carbon equivalent emitted
D = dollar damages form climate change, in the U. S. or the world
GDP = gross domestic (or world) product
subscript US = United States
subscript W = world

The ratio of U. S. GDP to world GDP (GDP_{US}/GDP_W) is about 25-30% today (Bureau of the Census, *Statistical Abstract of the United States,* 1992), but can be expected to decline slightly (over the period for which climate-change damage estimates are made) because the output of many large countries in Asia and South America will expand more rapidly than will U. S. output. I assume that over the future period for which global warming damages have been calculated, the quantity GDP_{US}/GDP_W will be between 0.20 and 0.28.

The quantity D/GDP is dollar damages from global warming per dollar of GDP. This quantity probably is lower for the U.S. (and for developed countries in general) than for the world, in part because the U. S. is wealthy enough to adapt relatively well to climate change, and in part because the U. S. might not suffer as severe effects as will some other countries (Tol, 1995, 1999, 2002b; Fankhauser, 1995; Nordhaus, 1994; Plambeck et al., 1997; IPCC, 1995c). For example, coastal flooding, droughts in agricultural regions, and deaths related to extreme weather probably will be less severe in the U. S. than in many other countries (especially undeveloped countries) Tol (1996) argues that both tangible and intangible damages are expected to grow with population and economic growth, but that as countries become more prosperous they should be able to reduce especially the loss of life, which is particularly costly. In one of his most recent model runs, Tol (2002b) finds that climate-change damages to America actually are *negative* (i.e., that climate change on the whole actually is beneficial) over about half of the period from 2000 to 2200.

However, if it turns out that the "intangible," non-market environmental damages are more severe than most analysts presently imagine, the imputed cost in the richer countries will be higher, because willingness to pay to avoid non-market damages (e.g., to natural ecosystems) generally rises with per-capita income (Plambeck et al., 1997, citing Manne et al.)

Fankhauser (1995), Tol (1995) and Nordhaus (1994) have estimated that D/GDP for the U. S. is 77% to 86% of D/GWP for the world. Similarly, the IPCC (1995c) reports that damages are expected to be 2%-9% of GDP in developing countries, 1% to 1.5% in developed countries, and 1.5% to 2.0% globally. (See also Plambeck et al., 1997.) However, more recent work by Tol (1999) implies that D/GDP for the U. S. is on the order of 15-50% of D/GWP for the world: he estimates marginal damages in North America of $1-3/t-C, versus marginal damages globally of $9-73/t-C. And Tol's (2002b) most recent work indicates that in North America the positive effects of climate change may about cancel the negative effects and result in marginal damages of approximately zero.

On the basis of these estimates, I assume that the ratio (D_{US}/GDP_{US}) : (D_W/GDP_W) is between 0.0 and 0.50.

With these assumptions, DT_{US} is equal to DPT_W multiplied by 0.0 to 0.14.

Total damages from emissions of greenhouse gases attributable to motor vehicles. As shown in Table 9-4 and Table 9-9, we estimate that emissions of greenhouse gases from motor-vehicle use in the U. S. cause less than $5 billion in damages in the U. S. The uncertainty in our estimate is due mainly to uncertainty in the marginal damage cost, which itself plausibly ranges over at least an order of magnitude. However, although more precise estimates of global warming damages await a better understanding of the consequences of global warming, the best available estimates indicate that motor-vehicle-related climate-change damages in the U. S. are relatively small compared with other motor-vehicle-related environmental damages.

9.2.9 The external cost of noise from motor vehicles (Report #14)

In many urban areas, noise can be a serious problem. Noise disturbs sleep, disrupts activities, hinders work, impedes learning, and causes stress (Linster, 1990). Indeed, surveys often find that noise is the most common disturbance in the home (Organization for Economic Cooperation and Development [OECD], 1988). And motor vehicles generally are the primary source of that noise (OECD, 1988).

In Report #14, we develop and document a model of the total external damage cost of motor-vehicle noise. We find that the external damage cost of direct motor-vehicle noise could range from less than $1 billion per year to nearly $50 billion per year, although we believe that the cost is not likely to much exceed $10 billion. In sensitivity analyses presented in Report #14, we show that this wide range is due primarily to uncertainty regarding the cost of noise per decibel above a threshold, the amount of noise attenuation due to ground cover and intervening structures, the threshold level below which damages are assumed to be zero, average traffic speeds, and the cost of noise outside of the home. Our estimates do not include the cost of "indirect" motor-vehicle noise, such as from highway construction, or the cost of controlling noise related to motor-vehicle use, or the loss of use of property that is unused because of motor-vehicle noise. Also, our estimates assume that motor vehicles are the only source of noise.

Our general cost model is conceptually straightforward: the dollar cost of noise is equal to the fraction of annualized housing value lost per excess decibel (HV), multiplied by: the annualized value of housing units exposed to motor-vehicle noise above a threshold (P); the density of housing units exposed to motor-vehicle noise above a threshold (M); the amount of motor-vehicle noise over a threshold (AN); and a scaling factor to account for costs in non-residential areas ((To+Ti)/Ti). Formally, we calculate the total cost of motor-vehicle noise in the U.S. with the following model:

$$Cn = \left(\sum_u \left(\sum_r \left(\sum_h AN_{u,r,h} \right) \right) \times M_u \times P_u \right) \times HV \times \frac{To + Ti}{Ti} \qquad \textbf{[9-16]}$$

$$AN_{u,r,h} = \frac{L_{u,r,h}}{5280} \cdot \left(\left(\int_{d_e}^{d_{t*}} Leq(d)_{u,r,h} \right) - ANB_{u,r,h} \right) \qquad \textbf{[9-17]}$$

$$Leq(d)_{u,r,h} = 10 \cdot \log \left\{ 0.0296 \cdot \frac{\Phi'}{180} \cdot V_{u,r,h} \cdot K_{u,r} \cdot \left(\frac{50}{d} \right)^{1+\alpha} \right\} - B_h \qquad \textbf{[9-18]}$$

where:

Cn = the total cost of motor-vehicle noise in the U.S. in 1990 (1991$)
$AN_{u,r,h}$ = the motor-vehicle "area-noise" level (we will explain this below) in area **u** along road type **r** with noise barrier of height-class **h** (zero height if no noise barrier) (dBA-mi^2)
$ANB_{u,r,h}$ = the motor-vehicle "area-noise" level below the noise-damage threshold t* in area **u** along road type **r** with noise barrier of height-class **h** (dBA-ft)
M_u = the density of housing units exposed to motor-vehicle noise above a threshold, in area **u** (number of housing units exposed to motor-vehicle noise above threshold t* divided by total land area exposed to motor-vehicle noise above threshold t* [units/mi^2])
P_u = the median annualized value of housing units exposed to motor-vehicle noise above a threshold, in area **u** ($/unit)
HV = the percentage of annualized housing value lost for each decibel of noise over the threshold level t*
Ti = the average amount of time spent in or around one's home (minutes)
To = the average amount of time spent away from one's home in places where motor-vehicle noise can be a problem (minutes)
$L_{u,r,h}$ = the total length of road type **r** in area **u** with noise barrier of height-class **h** (zero height if no noise barrier) (mi)
d_{t*} = the "equivalent distance" from the roadway to the point at which traffic noise drops to the threshold level (ft) ("equivalent distance" is defined below)
d_e = the "equivalent distance" from the roadway to the closest residence (ft) ("equivalent distance" is defined below)
t* = the threshold noise level below which the damage cost is presumed to be zero (dBA)

$Leq(d)_{u,r,h}$ = motor-vehicle noise (decibels) as a function of distance d from the road edge, for type of road **r** in area **u** with noise barrier of height-class **h**. This function is integrated from the point e, at the closest residences, up to the point at which the noise level drops off to the threshold level t*. The units of the integrated equation are dBA-ft. The equation is from the FHWA's Transportation Noise Model (Anderson, 1995)

5280 = feet/mile

Φ' = the "equivalent" subtending angle (the subtending angle is defined below), used to model the decrease in the noise level caused by intermediate obstructions (estimated using equations from Jung and Blaney, 1988)

$V_{u,r,h}$ = traffic volume (vehicles/hour) in urban area **u** on road type **r** with noise barrier of height class **h**.

$K_{u,r}$ = the total noise-energy emissions from different vehicle classes in urban area **u** on road type **r**

d = the "equivalent distance" in feet, equal to $\sqrt{dn \times df}$ where dn is the distance from the middle of the near lane to noise recipient, and df is the distance from the middle of the far lane to the noise recipient

α = the site parameter, or ground-cover coefficient (unitless); used to model the decrease in noise due to different types of ground cover

Φ = the subtending angle: the angle between two lines emanating towards the road from the noise receptor; one line drawn perpendicular to the axis of the roadway, the other drawn from the noise receptor to the edge of the obstruction (house, hill, etc.) along the roadway

B_h = the reduction in noise level provided by a sound wall of height-class **h** (zero height and zero reduction if no noise barrier) (dBA)

subscript u = area (377 urbanized areas plus 1 aggregated rural area; we use "u" rather than "a" because most of the areas are urbanized areas)

subscript r = type of road (the six types used by FHWA are: Interstate, Other Freeway, Principal Arterial, Minor Arterial, Collector, and Local)

subscript h = height class of noise barriers along the road (none, low, medium or high)

The model documentation in Report #14 presents the equations for α, Φ, and several other variables. In the following paragraphs we discuss the values for a few of the key parameters.

Simplifying assumptions underlying the motor-vehicle area-noise submodel
Although we account for a number of important factors, including traffic volume, traffic speed, the fraction of vehicles accelerating at any one time, the distance from the road, noise absorption by the ground, the angle defined by intermediate obstructions, and the extent and height of noise barriers, we also omit or simplify several important factors. For example, we assume that all vehicles travel on smooth, level roads -- we do not estimate the effects of rough roads and potholes. We do not include noise from horns,

sirens, skidding cars, or starting or revving engines. Although we do account for noise attenuation due to ground cover and intermediate obstructions, our treatment is crude.

The subtending angle (Φ). Houses, trees, hills, and other objects close to a road shield housing units further back from some of the road noise. The noise attenuation provided by this shielding depends on the location, size, height and other characteristics of the intervening "shields" and the shielded houses. The FHWA Traffic Noise Model includes a relatively sophisticated calculation of the attenuation due to shielding (Blaney, 1995). However, it is not possible to model shielding in detail in every area in the U.S. Instead, we adopt a much simpler approach, and use the subtending-angle parameter in the Jung and Blaney (1988) equation to model the effect of shielding.

In our formulation, the subtending angle is one-half the angle of sight framed by intervening objects. We assume in our base case that average "line of sight" to the road, or open noise path to the road, throughout an exposed residential area, is a sweep of 40 degrees, or 20 degrees on either side of the perpendicular, so that Φ=20. In sensitivity analyses, we test the effect of varying Φ from 10 degrees to 60 degrees. Interestingly, it turns out that noise costs are roughly proportional to the subtending angle, such that if the angle is doubled, costs roughly double.

The ground-cover coefficient (α). The ground-cover coefficient, α is a unitless number (between 0.0 and 1.0) meant to account for the noise attenuation caused by ground cover between the noise source and the receptor. On the basis of data in Jung and Blaney (1988), and recognizing that in large areas of central cities most of the ground is hard, we assume in our base case scenarios that α = 0.40. In sensitivity analyses, we test effect of varying α from 0.20 to 0.60.

The threshold noise level below which noise has no cost (t*). Noise damages are not continuous down to zero decibels. It is widely agreed that in most situations there is a nonzero threshold noise level below which most people will not be annoyed and above which most will be annoyed, although as the Organization for Economic Cooperation and Development (OECD, 1986) emphasizes, the threshold is different for different people and in different places. Our literature review (OECD, 1986; Linster, 1990; OECD, 1988; Rothengatter, 1990; Vainio, 1995) indicates that the threshold is about 55 dBA. Recently, Vainio (1995) tested "different partially linear noise specifications," and found that "the cutoff level of 55 dBA L_{eq} is supported by the data" (p. 163). In a scenario analysis, we test the effect of assuming a 50 rather than 55 dBA threshold, and find that damages roughly triple.

The diminution in annualized housing value per excess decibel (HV). Several studies (Nelson, 1978; Hall and Welland, 1987; O'Byrne et al., 1985) have estimated the shadow price of noise in the housing market by regressing sales price or property value against noise and other explanatory variables, such as lot size, number of rooms, and number of bathrooms. The estimated effect of noise on housing value is expressed as a percentage of value lost per decibel of noise above a threshold level. On the basis of these studies, we assume that each decibel of noise above a threshold reduces the value of a home by 0.2% to 1.0%. Note that the estimated noise costs are directly proportional to the value of this parameter.

Time spent in one's home, and outside of one's home (Ti, To). Traffic noise causes damages at places other than one's home or residential property. We account for these costs by extrapolating residential costs in proportion to the amount of time spent outside (To) versus in or around (Ti) one's home. To do this, we use a simple binary classification: in every away-from-home location, the exposure to and cost of motor-vehicle noise either is zero, or else is the same as the exposure to and cost of motor-vehicle noise in one's home, per minute on average. The basis of this classification is our judgment. For example, it seems reasonable to assume that motor-vehicle noise can be a problem in offices, schools, and churches, but not at nightclubs or shopping malls.

In an average day in California, people spend 921.1 minutes at home (Ti), and 250.6 minutes at non-home places (To) where in our judgment motor-vehicle noise might be a problem (based on data in Wiley et al., 1991). In a scenario analysis, we assume that motor-vehicle noise also disturbs the time spent in transit (111.4 minutes) and an additional 62.7 minutes of activities in various indoor and outdoor activities, bringing the parameter "To" to 424.7 minutes.

Results. Our scenario analyses indicate that motor vehicle noise could cost as little as $0.4 billion, or as much as nearly $50 billion. However, we view the high-end result as unlikely, and judge that noise costs probably do not exceed $10 or $20 billion annually. We present a range of $0.5 to $15 billion in Table 9-9. Our estimated range is consistent with the results of nearly 20 studies of the cost of traffic noise in Europe and the United States, from 1975 to 1991, reviewed by Verhoef (1994) and Rothengatter (1990), and with the earlier analysis for the U.S. by Fuller et al. (1983).

We also estimate the noise-cost per mile of travel by five different kinds of vehicles for six different kinds of roads, for a 10% increase in noise. The base-case, low-cost, and high-cost estimates from Report #14 are shown in Table 9-6 here. Note that the low and the high cost/mile estimates differ from the base-case estimates by more than an order of magnitude.

Note, too, that ours is an estimate of [most of the] external damage cost of direct noise from motor vehicles. This external cost, of course, is not the same as the total social cost of noise related to motor-vehicle use: the total social cost of noise related to motor-vehicle use is equal to the external damage cost of noise directly from motor-vehicles, which is what we have estimated here, *plus* the external damage cost of noise from "indirect" or "upstream" activities related to motor-vehicle use (such as highway construction), *plus* the cost of controlling noise related to motor-vehicle use. Moreover, as implied above, we have not counted every direct external cost of motor-vehicle noise: for example, we have not estimated all damages to property unused because of motor-vehicle noise.

Finally, we emphasize that we have estimated the cost of traffic noise as if traffic were the only major source of noise; we have not estimated the cost of traffic noise when there also is noise from, say, airplanes, trains, public events, or construction equipment. It is not possible to do a general, national analysis of the cost of motor-vehicle noise when there are other sources of noise, not only because it is not possible to identify and quantify all of the other noise sources, but because noise from one source does not add in a straightforward manner to noise from another source[22]. In some cases, an estimate of the noise contribution of motor-vehicles alone (i.e., as if there were no other noise source, which is our assumption) will underestimate the marginal contribution of motor vehicles when there are other sources, but in many cases, the reverse will be true. Consequently, we do not speculate about how an analysis of the cost of incremental motor-vehicle noise, given other sources of noise, might differ from our analysis. Also, we remind the reader that, as mentioned above, it does appear that traffic is the main source of noise in most people's lives.

9.2.10 Health and environmental impacts of leaking motor-fuel storage tanks

Some motor fuel leaks from underground storage tanks, contaminates groundwater, and causes health problems and property damage. Ideally, we would model these external costs of leaking storage tanks in several steps: characterize the population of underground tanks; estimate the probability of leaks of various sizes and types as a function of the characteristics of the tanks; model the dispersion and fate of fuel leaks; model the exposure of people and susceptible property to fuel leaks, based on the modeled fate of the leaks; estimate the effects of exposure to the fuel; and estimate the dollar value of any effects *not* covered by third-party liability insurance of tank owners. However, there are not enough data and modeling tools to be able to do this satisfactorily for the entire nation. (For a review of soil analysis, modeling, exposure assessment, site remediation, and related issues, see *Hydrocarbon Contaminated Soils and Groundwater*, 1991.)

We are forced, then, to supposition. As a starting point, we note that in 1987, DeLuchi et al. estimated that leaks from the underground tanks that store transportation fuels caused \$0.3 to \$1.0 billion in property damages, and \$0.3 to \$2.0 billion worth of health problems, including excess mortality. Their figure was based on estimates that in the early-to mid-1980s at least 100,000 tanks were leaking, and that a leaking storage tank could cause perhaps millions of dollars of property damage and threaten the groundwater supply of thousands of people.

Since the DeLuchi et al. (1987) guesstimate, however, the Environmental Protection Agency (EPA) has enacted strict regulations on leaking underground storage tanks (Federal Register, September 23, 1988; Federal Register, October 26, 1988). As explained below, I suspect that the new leakage prevention and clean-up programs have cut the external costs to much less than \$1 billion per year.

[22]The additive properties of two simultaneous noise sources depend upon their frequency structures. If the two noises are of wide frequency range and equal in intensity, they add in such a way as to increase the noise level by 3 dB. For two noise sources with a difference of 1 dB, the additive effect is to increase the louder noise by 2.5 dB. As the difference increases, the additive effect of the lower noise source becomes smaller, and when the difference in noise level reaches 10 dB, the louder noise source dominates the quieter one (Moore, 1978).

The UST program: technical and financial liability requirements. The EPA's Underground Storage Tank (UST) regulations establish technical requirements and financial liability requirements for operators of underground storage tanks. The technical requirements specify measures for new and existing underground tanks to prevent, detect, and clean up leaks. New tanks must meet strict design standards, and existing tanks must be fitted with release-detection systems (Federal Register, September 23, 1988). Tanks must be upgraded by 1998. The financial liability requirements are that tank operators with more than 10,000 gallons/month throughput must maintain financial assurance to cover at least $1,000,000 in mitigation and third-party liability costs per occurrence (Federal Register, October 26, 1988)

The UST regulations were costly to some operators. By now, however, most operators should have complied with the 1988 regulations, and henceforth the costs of the UST program will be small (*Environmental Investments, the Cost of a Clean Environment,* 1991, p. 5-5).

The LUST fund: money for clean up during the phase-in of the UST regulations. In 1986, before the UST technical and financial liability regulations, Congress authorized a fund to clean up leaking underground storage tanks:

> The purpose of the Leaking Underground Storage Tanks Program (LUST) is to ensure rapid and effective responses to releases from underground storage tanks containing petroleum and other hazardous substances. The program operates under the authority of Subtitle I of the Hazardous and Solid Waste Amendments of 1984, as amended by the Superfund Amendments and Reauthorization Act of 1986. The LUST program's objective is to enhance state and local enforcement and response through technical and financial assistance (EPA, 1993b, p. 14-5).

The LUST program fund was financed by a 1/10 of a cent per gallon tax on motor fuels. The fund was reauthorized by the Omnibus Budget Reconciliation Act of 1990 for an additional five years (EPA, 1993c, p. 55), and expired on January 1, 1996 (FHWA, 1997). It was reinstated on October 1, 1997, and will expire again on March 31, 2005 (FHWA, 1997).

The LUST fund provided clean-up funds during the phase in of the technical and financial liability requirements for storage tanks. A few years into the phase in, the leaking storage tank problem still was formidable: in 1993, the EPA claimed that "releases are being reported at a rate of 1,000 per week, and state workers are overseeing up to 400 cases at a time. Of the nearly 2 million regulated tanks, it is estimated that 20% may be leaking " (EPA, 1993b, p. 14-5). (About 50% of the regulated tanks contain motor-vehicle fuels.) However, the EPA also expected that the problem would be brought under control relatively rapidly:

> All state LUST programs have now developed response and enforcement capabilities. The Agency anticipates that the majority of leaks will be addressed by the states and responsible owners/operators. The state programs, supported by the Federal Trust Fund, will continue to emphasize finding responsible parties and performing the necessary compliance and enforcement work to get them to undertake corrective action at petroleum leak sites. (EPA, 1993b, p. 14-5).

> The EPA anticipates the overwhelming majority of these reported leaks will be addressed by the states and responsible owners/operators. In conjunction with the states, responsible parties have conducted almost 97% of all cleanups at leaking underground storage tanks. (EPA, 1993c, p. 56.).

The tank upgrades, the financial liability requirements, and the EPA's clean-up and prevention program undoubtedly have greatly reduced the frequency, magnitude and *external* cost of leaking storage tanks. (The private cost of preventing and cleaning up leaks presumably has been incorporated into the service-station margins in the price of gasoline.) It seems reasonable to assume that the external cost of leaking tanks has been cut by about an order of magnitude over the past decade, and now is significantly less than $1 billion per year.

Disposal of tanks. At the end of its life, an underground petroleum storage tank either is abandoned in place, or else removed and recycled or dumped in a landfill. Recycling probably is the most benign environmentally: the recycled material presumably does not enter the environment, and in any case the tanks usually must be thoroughly cleaned and conditioned before being recycled (Robinson et al., 1988). Properly abandoned or landfilled tanks also probably have relatively small environmental impacts.

However, improperly or illegally disposed tanks can cause environmental problems. If a tank is not properly dismantled and drained and either filled with inert material or else thoroughly cleaned, it probably will leak residual petroleum, sludge, lead, oxidized iron, and other toxic materials into groundwater (Robinson et al, 1988). Abandoned tanks -- especially those at or below the water table -- probably present the greatest risk. These leaks of hazardous materials from disposed storage tanks may present health and environmental threats similar to those from leaking in-service tanks.

Unfortunately, it appears that even less is known about the quantitative risk from disposed of tanks than about the risk from operating tanks. I speculate that the health and environmental costs of leaks from abandoned or landfilled tanks are no greater than costs of operating tanks, and probably are significantly smaller.

I assume, therefore, that the entire health and environmental cost of underground petroleum storage tanks, including costs of disposed tanks, is significantly less than $1 billion nationally at present (1997). However, as indicated above, the cost probably was much higher in 1991. In Table 9-9 I include what I assume to be the present cost, and in a footnote present what I assume to be the actual cost of the situation in 1991.

9.2.11 Environmental and economic impacts of large oil spills

Large oil spills can seriously disrupt marine ecosystems, and cause substantial economic losses to fisheries and tourist industries. They usually attract considerable attention, and sometimes engender new or tougher environmental laws[23].

There have been a number of estimates of the damage cost of large oil spills (see Behrens et al. [1992] and DeLuchi et al. [1987] for a review of some of the valuation studies). However, for several reasons, it is difficult to derive from these an estimate of the external cost of oils spills related to motor-vehicle use. In the first place, it is not clear to what extent oil spills are a cost of motor-vehicle use. Whether or not the oil spills that affect the U.S. (and note, we limit ourselves here to spills that affect the U. S.) are an avoidable cost of motor-vehicle use depends on how much and which crude-oil production changes as a result of a change in demand for transportation fuels. Thus, there are two key questions. First, How much does oil production change in response to a change in consumption of petroleum transportation fuels? The correspondence is not one-to-one, because petroleum is used in nontransportation sectors, and a change in demand for petroleum transportation fuels changes the price of all petroleum products and hence changes final demand for petroleum in nontransportation sectors.

The second question is: Which or whose oil production is affected? Oil spills are a marginal external cost of motor-vehicle use only if a change in motor-fuel use changes the frequency or severity of oil spills, and this will happen only if the affected oil comes from producers who ship to or from the U. S. by international tanker. In theory, a change in final demand for petroleum transportation fuels affects the marginal or high-cost oil[24]. If this marginal crude is transported by marine tankers, then there might be oil spills that affect the U.S.; if so, the spills can be considered to be a cost of motor-vehicle use in the U.S. (subject to consideration of the first question). But the marginal oil may just as well come from high-cost land-based stripper wells and be transported and stored on land exclusively, in which case there is no risk of marine oil spills at all. and no marginal oil-spill external cost of motor-vehicle use.

It is difficult to construct a model of the relationship between motor fuel use and the frequency and severity of oil spills. To my knowledge, it has not been done. This problem is particularly intractable because recent legislation -- the Oil Pollution Act of 1990 (OPA-1990) -- renders historical data, which one might have used as a basis of some kind of risk model, virtually useless. The OPA-1990 requires that oil tankers be double-hulled, and that tanker owners and operators be financially capable of paying for oil clean-up (FHWA et al., 1996):

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[23]For example, in response to the oil spill -- the largest in U. S. history -- from the grounding of the *Exxon Valdez* in Prince William Sound on March 24th 1989, Congress passed the Oil Pollution Act of 1990 (OPA-1990) (discussed more below).

[24]In reality, the decision about which oil to produce is not determined solely by short-run or long-run marginal costs, but also is influenced by contractual obligations, national laws, and political factors.

• all non-double-hull tankers serving U. S. ports be phased out beginning in 1995, and completely by 2010 (single-hull tankers) or 2015 (double-bottom or double-sided tankers);

• all newly built tankers delivered after 1993 be double-hulled;

• tanker owners and operators who load or discharge at U. S. ports maintain a Certificate of Financial Responsibility that shows that they are capable of paying for oil cleanup operations;

• in Prince William Sound and Puget Sound (and now perhaps elsewhere), laden single-hull tankers be escorted by tugboats.

These regulations, which increase the market cost of transporting oil[25]. certainly will decrease the *external* cost of oil spills.

We do not formally address these questions here. Instead, we simply review the discussion, data, and estimates in reports by the California Energy Commission (Stevens and Peterson, 1993), the Congressional Research Service (Behrens et al., 1992), and DeLuchi et al. (1987), and estimate a present annual-average oil spill cost of something on the order of $0.10/barrel of oil produced. Given this, we estimate the range of motor-vehicle cost shown in Table 9-9.

9.2.12 Other water pollution related to motor-vehicle use: urban runoff polluted by oil from motor vehicles, and nitrogen deposition

Urban runoff. Oil, fuel, coolant, and other chemicals leak or are discarded from motor vehicles, and eventually pollute rivers, lakes, wetlands, and oceans.

Recent research has shown that motor vehicles are a major source of pollution in urban runoff. In a study of the sources of petroleum hydrocarbons in urban runoff from highway, industrial, residential, and commercial sites in Rhode Island, Latimer et al.

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[25]According to the FHWA et al. (1996), an economist at Lloyd's Shipping estimates that double-hull tankers cost 15% to 20% more than comparable single-hull tankers. These extra costs presumably are incorporated into the price of oil.

This brings up the question of how to handle the oil-spill liability tax. In Report #17, we note that in 1991, domestically produced crude oil and all imported petroleum was assessed an environmental excise tax of $0.05/bbl for the Oil Spill Liability Trust Fund, which was established to prevent and clean up oil spills and to compensate individuals for damages caused by oil spills. (The tax is to be suspended when the fund accumulates $1 billion, which occurred on July 1, 1993 [Barthold, 1994].) These taxes were imposed on domestically produced crude oil upon receipt at the refinery and on imported petroleum when it entered the United States (Internal Revenue Service, *Excise Taxes for 1994*, 1993). Now, if this excise tax were a true Pigovian tax, equal to marginal expected damages of oil spills, then it would make most sense to count the damages as part of the private cost of fuel, by letting the tax remain in the calculated cost of the fuel. Then, there would be no *external* cost of oil spills. However, we doubt very much that the $0.05/bbl excise tax is a correct Pigovian tax. If it is not, then the correct approach is to treat the excise tax as a general tax and eliminate it from the estimate of the private cost of fuel, and to calculate the environmental damage cost as an externality. So, in Report #5, we deduct from the calculated cost of fuel the amount of the Oil Spill excise tax embedded in the cost, and in this report, estimate the external damage cost.

(1990) found that the distribution of polycyclic aromatic hydrocarbons in the runoff was quite similar to the distribution in used crankcase oil. On the basis of this, they conclude that most of the hydrocarbons in urban runoff come from used crank case oil. Similarly, a multi-year, intensive study by FHWA determined that "deposition from vehicles is the primary source of pollutants except during periods of ice and snow, when deicing chemicals and abrasives are the primary source" (Bureau of Transportation Statistics, 1996, p. 149)[26]. Motor-vehicles, motor-fuels, lubricants, tires, brakes, roadbeds, engine parts, and other sources related to motor-vehicle use were found to be primary contributors to particulate, petroleum, and metal loadings in highway runoff (Smith and Lord, 1990). In support of this, Bannerman et al. (1993) found that streets, driveways, and parking lots generate 80 to 95% of the metal pollutants in storm runoff (lawns, roofs, and sidewalks contribute the rest). And finally, according to the EPA (1993a), motor-vehicle use is responsible for most of the organic pollutants in storm runoff as well.

This polluted runoff, in turn, can significantly degrade rivers, lakes, streams, and wetlands. Storm runoff affects 11% of the polluted river-miles, 28% of the polluted lake-acres, 30% of the polluted estuary square miles, 36% of polluted ocean coastal miles, and 2% of polluted wetland-acres in the U. S. (EPA, 1992).

Unfortunately, there is not enough information available yet to get from these general findings to a real model of the dollar cost of urban runoff due to motor vehicles. I speculate that the cost of this pollution is comparable to the cost of groundwater pollution from leaking underground storage tanks, and hence assume a cost of $0.1 to $0.5 billion per year (Table 9-9). (Note that this is the cost of the environmental damage; the motor-vehicle-related cost of the sewer-system pollution control is counted as a government cost, in Report #7.)

Nitrogen deposition. A substantial fraction of the nitrogen emitted from motor vehicles (as NO_X) deposits out of the atmosphere onto soil, plants, man-made structures, and water bodies. This nitrogen deposition can stress plants, corrode materials, and eutrophy bays and lakes by feeding algal blooms that consume oxygen as they decay. In the Chesapeake Bay, air pollution accounts for some 27% of the total nitrogen load (E. H. Pechan Associates, 1996).

Unfortunately, there are insufficient data to estimate either the contribution of motor-vehicle NO_X emissions to water degradation nationally, or the cost of that degradation[27].

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[26]The FHWA study of highway runoff spanned more than15 years, ending in 1989, and produced more than 20 reports: 6 reports on the *Constituents of Highway Runoff*, 4 reports on *Sources and Migration of Highway Runoff Pollutants,* 5 reports on the *Effects of Highway Runoff on Receiving Waters,* 4 reports on the *Evaluation of Pollutant Impacts from Highway Stormwater Runoff,* and 3 reports on *Retention, Detention, and Overland Flow for Pollutant Removal from Highway Stormwater Runoff* (Smith and Lord, 1990). Summaries, guidelines, and reports on related topics, some by other agencies, also were produced.

[27]E. H. Pechan Associates (1996) report the results of an analysis of the cost effectiveness of reducing nitrogen loads to the Chesapeake Bay by controlling regional air emissions of NO_X, but the it is

9.2.13 Nonmonetary costs due to net crimes related to using or having motor-vehicle goods, services, or infrastructure

Crime imposes several kinds of costs on society: police enforcement, individual defensive behavior, direct monetary losses of victims[28], pain and suffering of victims, and more. The pain and suffering is an external cost, and can and should be valued and counted in a social cost analysis that includes crimes, just as the pain and suffering from motor-vehicle accidents is counted. For serious crimes, the pain and suffering cost exceeds the direct economic cost. For example, Kleiman and Cavanagh (1990) estimate that the "social costs" (including pain and suffering) of crime, beyond the direct economic costs to victims, are 250% to 350% of the direct economic costs.

The cost is estimated simply as the number of victimizations multiplied by the cost per victimization, with allowance for the possibility that there are substitutes for many crimes:

$$PSC = \frac{\sum_{C} VCT_C \cdot (PS_C + LP_C) \cdot SUBF_C}{1000000} \qquad \textbf{[9-19]}$$

where:

PSC = the cost of pain, suffering, death, and lost quality of life caused by motor-vehicle related crime (10^6 \$)
VCT_C = number of victimizations of type C (Table 9-7)
PS_C = the pain and suffering cost per victimization of type c (\$/victimization; Table 9-7)
LP_C = the cost of non-monetary lost productivity, per victimization of type c (\$/victimization; Table 9-7)
$SUBF_C$ = the fraction of VCT_C remaining after substitute crimes are accounted for (Table 9-7; see also brief discussion below and in Report #7).
subscript c = types of motor-vehicle-related crime (see Table 9-7)

As discussed in Report #2, and in the notes to Table 9-7, our estimate allows that some crimes that apparently are related to motor-vehicle use actually are opportunity costs not of motor-vehicle use per se, but rather of the possibility for criminal behavior in general. That is, our estimates allow that in many cases *other* criminal opportunities

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impossible to extend the their analysis (which in any case considered control costs, not damage costs) to the whole nation.

[28]To the extent that they are not merely forced transfers, the direct monetary losses, such as the value of stolen vehicles -- already are included in our estimates of payments for motor-vehicles, parts, accessories.

will substitute for opportunities apparently related to motor-vehicle use. Table 9-7 shows estimates of the fraction of the total number of instances of each general type of offense (murder, robbery, theft, arson) that would in fact be eliminated altogether were there no motor-vehicle use -- in other words, the fraction remaining *after* substitution of other crimes for nominally "motor-vehicle"-related crimes. For example, a weight of 20% on gas-station robberies means that we assume that were there no motor vehicles (and hence no gas stations), the total number of robberies of all kinds would be reduced by only 20% of the number of gas-station robberies (i.e., we assume that there are substitutes for 80% of gas station robberies). Thus, only 20% of gas station robberies would be an opportunity cost of motor-vehicle use.

The data in Table 9-7 come mainly from Miller et al. (1995). The unit cost estimates of Miller et al. (1995), and the similar estimates of Cohen (1988), are based on jury awards to victims. As a result, they do not account for the cost of fear, anxiety, and avoidance behavior of those who have not yet been victimized. However, we know of no way to estimate this.

Note that I do not count here the cost of the injuries and fatalities to police personnel, because in theory, the risk of injury or death is represented by a wage premium demanded by police personnel, or else is covered by the life insurance provided them as part of their compensation. Similarly, the monetary costs of hospital stays by police personnel are covered by their medical benefits, and the monetary costs of accidental death may be covered by their life insurance. Thus, the cost of injury and death to police personnel should be included in their compensation, and hence included already in my estimates in Report #7 (Table 7-1) of government expenditures on police protection[29].

As discussed in Report #2, we have classified the costs of motor-vehicle related crime as "externalities" in order to keep all costs neatly within our economic-efficiency framework, not because we believe that one necessarily should take an a-moral view of crime, or somehow address the "externalities" of crime solely through some kind of optimal taxation (!). To the contrary, it is more sensible to classify illegal and immoral behavior as precisely that, and to address the problems by law enforcement and moral suasion rather than by economic incentives.

9.2.14 Pain, suffering, and other nonmonetary costs due to fires related to using or having motor-vehicle goods, services, or infrastructure

I estimate the cost of pain, suffering, death, and lost quality of life due to motor-vehicle-related fires as the product of: fire-related injuries or deaths to civilians, and the non-monetary cost of injuries or deaths. I do not count here the cost of injuries or fatalities to fire personnel for the same reason that I do not count here the injuries or

[29]Note, too, that non-monetary cost of arson to motor-vehicles and arson to gas stations and car dealerships is counted elsewhere as a cost of fires related to motor-vehicle use, and the non-monetary cost of driving under the influence and hit and run is counted elsewhere as a cost of motor-vehicle accidents.

deaths to police personnel: the costs are included in the compensation of fire personnel and hence in my estimates in Report #7 (Table 7-1) of government expenditures on fire protection[30].

Formally:

$$PSDF = \left(RPF \cdot F_{CR} + \sum_{F} F_F \cdot MV_F \right) \cdot VOL \cdot NMF + \left(RPI \cdot I_{CR} + \sum_{F} I_F \cdot MV_F \right) \cdot COI \cdot NMI$$

[9-20]

where:

PSDF = the cost of pain, suffering, death, and lost quality of life caused by motor-vehicle related fires (10^9 $)

RPF = civilian fatalities resulting from fires on roadway property (Table 7-12)

F_{CR} = of RPF, the fraction that is not counted already as a fatality due to a motor-vehicle crashes (I assume 0.05 to 0.10, because it appears that most fatal fires on "roadway property" are the result of an accident [based on an analysis of fire incidents reported by the U. S. Fire Administration, 1992])[31]

F_F = civilian fatalities from fires on property type F (Table 7-12)

MV_F = of fires on property type F, the fraction that is related to motor-vehicle use (Table 7-12)

VOL = the total (monetary plus nonmonetary) value of life ($1.0 to $4.0 million; Report #11)

NMF = the nonmonetary fraction of the total cost of a fatality (0.69; based on Miller, 1997 and Blincoe, 1996; includes lost household productivity as well as pain and suffering)

RPI = civilian injuries resulting from fires to roadway property (Table 7-12;

I_{CR} = of RPI, the fraction that is not counted already as an injury due to a motor-vehicle crash (see F_{CR})

I_F = civilian injuries from fires on property type F (Table 7-12)

COI = the total cost (monetary plus nonmonetary) of a generic civilian injury in a motor-vehicle related fire (I assume $50,000 to $150,000, on the basis of data discussed in the next section)

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[30]Note that, in any case, there are far, far fewer injuries and deaths to fire personnel than to civilians.

[31]Fatalities and injuries due to fires in motor-vehicle crashes are included in the fatality count for motor-vehicle crashes (NHTSA, 1993).

NMI = the nonmonetary fraction of the total cost of an injury (0.67; based on Miller, 1997 and Blincoe, 1996; includes lost household productivity as well as pain and suffering)

The total cost of a generic civilian injury (COI). Miller et al. (1995) estimate that an arson injury costs $200,000, and Miller (1997) estimates that the average non-fatal injury in from a motor-vehicle crash costs $45,470 in 1995 dollars. Blincoe and Faigin (1992) and Blincoe (1996) also estimate the "comprehensive" costs (medical costs plus pain and suffering costs) of injuries classified according to the "Maximum Abbreviated Injury Scale (MAIS):

MAIS	description (example) (from Miller et al., 1991)	Blincoe and Faigin (1992) 1990 $	Blincoe (1996) 1994 $
1	minor (1st-degree burn)	9,231	10,840
2	moderate (major abrasion)	110,771	133,700
3	serious (multiple rib fracture)	405,573	472,290
4	severe (spleen rupture)	1,024,871	1,193,860
5	critical (spinal cord injury)	2,219,892	2,509,310
6	fatal	2,628,961	2,854,500

These estimates suggest that a range of $50,000 to $150,000 is appropriate for a generic civilian injury.

9.3 EXTERNAL COSTS NOT ESTIMATED IN THIS REPORT

As indicated in Table 9-9, we have not estimated several of the externalities of motor-vehicle use: air pollution damages to ecosystems other than forests, the environmental and esthetic impacts of motor-vehicle waste, vibration damages from motor-vehicles, and fear and avoidance of motor vehicles and crimes related to motor-vehicle use. I emphasize that this is because we are not able to make even remotely credible estimates, not because we believe the costs to be trivial and not worth estimating. I discuss one of these costs, motor-vehicle waste, briefly in the next section.

9.3.1 Environmental and esthetic impacts of motor-vehicle waste

The production, usage, and disposal of motor-vehicles and fuels produces a variety of solid wastes, including dredge from resource extraction, waste sludge from oil refineries, excess material from vehicle manufacture, and of course scrapped vehicles and parts.

Undoubtedly, scrappage is the most important source of solid waste in the motor-vehicle production-and-use system. Presumably, most of the waste of the motor-vehicle system -- cars, car parts, tires, batteries, debris from highway construction, and more -- is properly disposed in landfills. Some of it, however, is dumped or abandoned illegally. This improperly disposed waste is always an eyesore, and can be dangerous and damaging to human health and ecosystems. Unfortunately, we are unable to estimate these esthetic and environmental solid-waste costs of the motor-vehicle use system. However, we do make a very crude estimate of the share of the cost of landfills that is attributable to motor-vehicle use (Report #7).

9.3.2 Fear and avoidance of motor-vehicles

Evans (1994) notes properly that "road safety externalities...include... the opportunity cost of actions taken by other road users to avoid accidents, for example by reducing their walking" (p. 5). Similarly, Newbery (1998) argues that "the fact that children are not allowed to play in streets and are driven to school rather than walking lowers the child accident rate but has a social cost" (p. 3). Although I am unable to estimate this cost, I speculate that it might not be trivial. For example, if the fear and avoidance cost is only 5% of the estimated total pain and suffering cost of motor-vehicle accidents, it still is on the order of a few billion dollars per year. Alternatively, if every household in the U. S. were willing to pay $50/year to not be frightened by motor vehicles[32], the national total would again be several billion per year.

9.4 NON-MONETARY ENVIRONMENTAL AND SOCIAL IMPACTS OF HIGHWAY INFRASTRUCTURE (NOT ESTIMATED IN THIS REPORT)

The construction, existence, and to some extent maintenance of the motor-vehicle infrastructure, apart from the level of use of the infrastructure, can destroy natural habitat, pollute water bodies, and divide communities. Moreover, the motor-vehicle infrastructure usually is unsightly.

To the extent that these costs are independent of marginal changes in motor-vehicle travel, it is inappropriate to call them externalities of motor-vehicle use. However, they clearly are social costs, and should be counted in cost-benefit evaluation of transportation policies and investments that affect the motor-vehicle infrastructure. In the following sections, I discuss these environmental and social impacts of the motor-vehicle infrastructure briefly, and qualitatively. Although I am unable to estimate the dollar cost, one should not necessarily presume that the costs are trivial.

[32] Note that this would be a payment not to change the expected costs of accidents, but rather to be free of the fear given the same expected accident costs.

9.4.1 Habitat destruction, and effects on plants and animals (not estimated in this report)

Road construction can fragment sensitive environmental habitat and thereby disturb and possibly even eliminate plants and other (non-human) animals. Van Bohemen (1995) distinguishes four kinds of fragmentation (p. 133):

- *Destruction:* absolute loss of a habitat area through the physical presence of the road and associated infrastructure;
- *Disturbance:* deterioration of the habitat due to traffic noise, pollution, lighting, and so on;
- *Barrier action:* separation of functional areas of habitat;
- *Collisions:* injury or death to animals by cars[33]

De Santo and Smith (1993) provide a similar typology of impacts: displacement of land, fragmentation of habitat, change in perimeter to area ratio of habitat, and obstruction of movement.

How pervasive are these impacts? A recent survey of 45 state transportation departments found that most of the states had communities of federally or state-protected plant communities within their highway right-of-way (Hanson and Hummer, 1994). The same survey found that most states have had projects delayed or redesigned because of endangered species, and that 9 states have had projects stopped by endangered species.

Generally, states are required to mitigate the adverse environmental impacts of highway projects. There are many ways to mitigate impacts: use special construction and maintenance techniques; relocate endangered species; avoid building during certain seasons of the year; avoid building in certain areas; protect sensitive areas; restore land elsewhere to compensate for land taken for the highway project; and more (Hanson and Hummer, 1994). For example, the elimination of 77 acres of freshwater wetlands by the construction of 20 miles of Interstate 287 in New Jersey is being mitigated by creating elsewhere “high value wildlife habitat and flood storage compensation at a 1:1 impact to mitigation ratio” (Fekete et al., 1994, p. 64).

Of course, mitigation does not always completely compensate for or eliminate the negative impacts of roadway development. Inevitably, roads will change the migration, feeding, nesting, and mating patterns of some animals (see for example McCartney et al., 1994).

Although there are studies of some kinds of costs of some kinds of impacts of specific kinds of habitats and ecosystems, there are insufficient data to make a national estimate of the cost of all habitat and ecosystem damage wrought by highways.

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[33]Cook and Dagget (1995) discuss “road kill” and related issues.

However, some studies of the value of wetlands suggest that the cost of the environmental impacts of highways may not be completely trivial. For example, Creel and Loomis (1992) developed a multinomial logit model of recreation site choice and trip frequency to estimate the economic benefit to hunters, anglers, and wildlife viewers of 14 wildlife areas and rivers in the San Joaquin Valley of California. They found that at the existing level of water quality, the total annual benefit of the 14 sites was $100 million to $1 billion (1989 $), depending on whether the relevant population pool of hunters, anglers and viewers was taken to be residents of the San Joaquin Valley only, or residents of the entire state of California. This implies something on the order of $100 per household per year. Whitehead (1990) used a contingent valuation survey to estimate the benefits of preserving a wetlands in Kentucky, and found that households were willing to pay between $6 and $13 per year (1989$) into a hypothetical Wetland Preservation Fund, with a total willingness to pay of $3 to $20 million to preserve the one wetlands in question.

Recently, Noland and Apogee Research (1997) estimated the cost of restoring wetlands lost to roads built with federal aid between 1955 and 1980. They multiplied the number of acres covered by federal-aid roads (3.2 to 7.7 million, depending on assumptions regarding the width of the roads) by a 5.76% chance that any acre covered a wetlands (assuming that roads were randomly located with respect to wetlands), then added 123,000 wetland acres lost because of indirect effects, and finally multiplied the resultant total acreage lost (307,000 to 572,000)[34] by unit replacement costs of $500 to $10,480 acre[35]. The result was a cost of $150 to $6 billion.

Willis et al. (1998) review studies of the "wildlife value" and "landscape value" of land used for roads in Britain. They report a very wide range of values, from less than £ 10/ha/yr to more than £ 10000/ha/yr, depending, naturally, on the type of land (forest, meadow, farm, etc.), and the type of values solicited (use value, option value, existence value, etc.). If, for illustrative purposes, one assumes that half of the road space in the U. S. has essentially no non-market amenity value, and the other half has a value of $ 100/ha/yr, the total non-market amenity value is on the order of $600

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[34]We can offer a similar calculation. In Report #6, we estimate that the entire motor-vehicle infrastructure -- paved and unpaved roads, parking lots, garages, driveways, loading docks, service stations, parts stores, and car dealerships -- covers some 25,000 square miles, or 16 million acres. However, this includes only areas actually covered by pavement, buildings, or graded dirt; it does not include land that is disturbed by the road construction but not paved or graded flat. Including such additional disturbed land might bring the total to 20 million acres. (Note that our figure includes the entire present infrastructure in place, whereas the Noland and Apogee Research [1997] figure includes only federal-aid roads built between 1955 and 1980). Multiplying by 5% (slightly lower than the 5.76% of Noland and Apogee, because of environmental laws enacted after 1980) results in an even 1 million wetland acres lost.

[35]The analyses of Creel and Loomis (1992) and Whitehead (1990) suggest that the *annual* willingness to pay to preserve wetlands ranges from less than $1,000/acre to well over $10,000/acre. This exceeds by a large margin the one-time restoration costs cited by Noland and Apogee Research (1997).

million per year (based on roughly 6 million ha of paved and unpaved public road in the U. S. [Report #6]).

On the basis of the foregoing, I suspect that the total willingness to pay to eliminate the negative impacts of highways on plants and animals is small compared to the air pollution externalities estimated here, but not necessarily trivial. In particular cases, however, the impacts might be quite large. Clearly, more research in this area is warranted.

9.4.2 The water-quality impacts of highway deicing.

Throughout the U.S., but primarily in the Northeast, rock salt is applied to highways to melt snow and ice. Although the salt is cheap and effective, it disintegrates pavement, corrodes vehicles and bridges, and pollutes groundwater (EPA, 1996). The polluted groundwater, in turn, can harm vegetation, wildlife, and, by increasing the salinity of drinking water, even humans (Murray and Ernst, 1976; EPA, 1993a).

In 1976, the EPA (Murray and Ernst, 1976) published a comprehensive economic analysis of the environmental impacts of highway deicing. They estimated the following damages and costs in billions of 1973-74 dollars:

i) pollution of water supply and damage to human health	0.150
ii) damage to vegetation	0.050
iii) corrosion of highway structures	0.500
iv) corrosion of motor vehicles	2.000
v) corrosion of water, telephone, and power lines	0.010
vi) purchase and application of salt	0.200
Total	2.910

The cost of salt corrosion of vehicles and highways (items iii and iv) already is included in our estimates, in Report #5 and #7, of the total cost of maintaining and replacing motor vehicles and highways. Hence, in this analysis, it is not an additional cost. The cost of purchasing and applying salt (item vi) also is included in our estimates of the cost of maintaining highways (Report #7). However, the damages to human health and vegetation (items i and ii), and the cost of corrosion of utilities (item v), are not included elsewhere, and so should be included here.

Items i, ii, and v amounted to $0.210 billion per year, in 1973-1974 dollars. Converting to 1991$, and dividing by the 9 million tons of salt applied in 1973-74 (Murray and Ernst, 1976) yields a figure of $60/ton applied (in 1991$). Because the EPA considered its estimates of the environmental costs to be lower bounds, we will assume $60/ton in our low-cost case, and twice this figure ($120/ton) in our high-cost case. In recent years, about 10 million tons of salt have been applied annually (EPA, 1996).

9.4.3 The socially divisive effect of roads as barriers

Poorly designed and thoughtlessly placed roads and freeways can divide communities, impede circulation, and create barriers to social interaction[36]. Indeed, the "freeway revolts" that began in the late 1960s and shut down freeway projects in several cities (the dead-end Embarcadero Freeway in San Francisco, torn down after the 1989 Loma Prieta earthquake, was perhaps the most famous example) were spawned in part by these sorts of negative social impacts. Soguel (1995) cites a study by Appleyard that shows that "residents of San Francisco with light volumes of traffic have three times as many local friends and twice as many acquaintances as those on heavily traveled streets" (p. 302)[37].

Soguel (1995) used a contingent valuation survey to estimate the cost of roads as barriers to easy and safe pedestrian access in the Swiss town of Neuchâtel (population 32,000). The residents were asked how much they would be willing to have traffic on five urban streets, for a total of 750 meters, diverted to underground bypasses. Presently, the streets are in the center of the city, and impede access to the main municipal garden, and a nearby recreational lake. The residents were willing to pay $1.9 to $2.6 million per year for the underground bypasses. This results in the sizable sum of $58 to $82/person/year.

Of course, for a number of reasons, one should not simply multiply these results for Neuchâtel by the number of people in cities in the U.S, and call the result the cost of roads as barriers in the U.S. In the first place, the residents of Neuchâtel paid hypothetically for underground bypasses, which would reduce the noise, pollution, and visual impact as well as the danger and impediment of motor vehicles. There is no way to know how people valued the barrier effect itself. Second, the roads in Neuchâtel might be unusually disruptive, because they impede access to the main municipal garden, and to a nearby recreational lake. If so, then the stated WTP overstates the WTP to mitigate the "average" barrier effect. Third, people in Swiss cities might be willing to pay more for a given improvement than people in U. S. cities. For these reasons, we cannot infer from the Soguel (1995) study the cost of the barrier effect in the U.S. I note, though, that it is possible that this cost is not trivial.

9.4.4 The esthetics of roads and the motor-vehicle service infrastructure

The motor-vehicle infrastructure also can be ugly (Button, 1993). Roads, gas stations, car lots, car-repair shops, parts stores, parking lots, and garages form dreary, chaotic strip developments decried by architects and city planners (e.g. Kunstler, 1993). Of course, one need not be esthetically sensitive by profession to be offended: surveys

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[36]Of course, roads also facilitate social interaction and provide access to goods, services, and amenities -- that in fact is their primary and very valuable function. That the main user benefit of roads is social and economic access does not mean that roads can not have undesirable external side effects on communities.

[37]This, of course, indicates that some of the divisiveness is due to incremental motor-vehicle use rather than to the mere existence of the infrastructure. To the extent that this is so, there is an externality of motor-vehicle use.

have found, not unexpectedly, that the general public feels that the world would be prettier without roads (Huddart, 1978). And the unsightliness of scrapped autos and junkyards has been formally condemned by the courts: according to Woodbury (1987), a court in Colorado ruled that a stockpile of old cars, scrap metal, petrochemical drums, and other obnoxious debris near a mountain cabin was an unsightly eyesore and had to be removed -- solely because it was unesthetic.

Partly as a result of the protests of the 1960s, the highway-planning process in the U.S. was amended to better accommodate community concerns, and it is unlikely that future highway projects will be as disruptive as were some of the past. However, by and large, urban planners and architects have not been able to control strip development and improve the esthetics of roadside America. We expect that motor-vehicles and the infrastructure that supports them will continue to exact an esthetic "cost" for the foreseeable future. However, even though researchers have been able to some extent to quantify the visual intrusiveness of roads (Grigg and Huddart, 1978), we are unable to estimate the dollar cost of the overall ugliness of the motor-vehicle infrastructure.

9.5 SUMMARY OF THE EXTERNAL COSTS

The estimates developed above are summarized in Table 9-9. The costs per kg of air pollution are summarized in Table 9-8.

Note that I have classified the nonmonetary social and environmental impacts of the motor-vehicle infrastructure separately from the non-monetary externalities of motor-vehicle use. Although these infrastructure costs ultimately are a long-run cost of total motor-vehicle use, they are not a cost of marginal or incremental motor-vehicle use, because they do not vary with each mile or trip. Hence, as noted above, infrastructure costs are not externalities of motor-vehicle use, according to our definition.

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TABLE 9-1A. THE HEALTH COST PER KG OF MOTOR-VEHICLE EMISSIONS, BASED ON A 10% REDUCTION IN DIRECT MOTOR-VEHICLE EMISSIONS (1990 EMISSIONS, 1991$/KG-EMITTED)

Emission	Ambient pollutant	United States		All urban areas		Los Angeles	
		low	*high*	*low*	*high*	*low*	*high*
CO	CO	0.01	0.09	0.01	0.10	0.03	0.18
NO_x	nitrate PM_{10}	1.02	16.56	1.39	22.38	6.05	75.83
	NO_2	0.15	0.73	0.19	0.96	0.52	2.64
Total for NOx		1.17	17.29	1.59	23.34	6.58	78.47
$PM_{2.5}$	$PM_{2.5}$	10.42	159.19	14.81	225.36	63.98	779.13
$PM_{2.5-10}$	$PM_{2.5-10}$	6.70	17.68	9.09	23.89	38.12	78.34
Total for PM_{10}		9.75	133.78	13.74	187.48	58.79	638.33
SO_x	sulfate PM_{10}	6.90	65.52	9.62	90.94	34.98	226.89
VOC	organic PM_{10}	0.10	1.15	0.13	1.45	0.51	4.34
VOC+NO_x	ozone	0.01	0.11	0.02	0.14	0.05	0.40

Source: Report #11.

Each $/kg value is equal to the total calculated health damages attributable to the pollutant and source, divided by emissions of the pollutant from the source.

TABLE 9-1B. THE HEALTH COST PER KG OF MOTOR-VEHICLE AND RELATED UPSTREAM EMISSIONS, BASED ON A 10% REDUCTION IN DIRECT MOTOR-VEHICLE EMISSIONS AND RELATED UPSTREAM EMISSIONS (1990 EMISSIONS, 1991$/KG-EMITTED)

Emission	Ambient pollutant	United States		All urban areas		Los Angeles	
		low	*high*	*low*	*high*	*low*	*high*
CO	CO	0.01	0.09	0.01	0.10	0.03	0.18
NO_x	nitrate PM_{10}	0.96	15.53	1.31	21.17	6.02	75.11
	NO_2	0.14	0.68	0.18	0.91	0.52	2.62
Total for NOx		1.10	16.21	1.50	22.08	6.54	77.73
$PM_{2.5}$	$PM_{2.5}$	9.71	147.24	13.63	205.44	62.57	737.36
$PM_{2.5\text{-}10}$	$PM_{2.5\text{-}10}$	5.30	14.25	7.20	18.34	30.04	47.87
Total for PM_{10}		8.78	116.01	12.23	158.23	54.68	509.18
SO_x	sulfate PM_{10}	2.80	22.60	4.40	35.28	33.53	209.88
VOC	organic PM_{10}	0.10	0.99	0.13	1.25	0.52	4.29
VOC+NO_x	ozone	0.01	0.10	0.02	0.12	0.05	0.39

Source: Report #11.

Each $/kg value is equal to the total calculated health damages attributable to the pollutant and source, divided by emissions of the pollutant from the source.

TABLE 9-1C. THE HEALTH COST PER KG OF MOTOR-VEHICLE AND RELATED UPSTREAM AND ROAD-DUST EMISSIONS, BASED ON A 10% REDUCTION IN DIRECT MOTOR-VEHICLE EMISSIONS, RELATED UPSTREAM EMISSIONS, AND PAVED-ROAD-DUST EMISSIONS (1990 EMISSIONS, 1991$/KG-EMITTED)

Emission	**Ambient pollutant**	**United States**		**All urban areas**		**Los Angeles**	
		low	*high*	*low*	*high*	*low*	*high*
CO	CO	0.01	0.09	0.01	0.10	0.03	0.18
NO_x	nitrate PM_{10}	0.96	15.53	1.31	21.17	6.02	75.11
	NO_2	0.14	0.68	0.18	0.91	0.52	2.62
Total for NOx		1.10	16.21	1.50	22.08	6.54	77.73
$PM_{2.5}$	$PM_{2.5}$	7.48	94.45	10.47	130.89	45.44	430.10
$PM_{2.5-10}$	$PM_{2.5-10}$	1.03	7.58	1.42	10.34	5.62	32.26
Total for PM_{10}		2.84	39.87	3.92	54.64	15.20	169.45
SO_x	sulfate PM_{10}	2.80	22.60	4.40	35.28	33.53	209.88
VOC	organic PM_{10}	0.10	0.99	0.13	1.25	0.52	4.29
VOC+NO_x	ozone	0.01	0.10	0.02	0.12	0.05	0.39

Source: Report #11.

Each $/kg value is equal to the total calculated health damages attributable to the pollutant and source, divided by emissions of the pollutant from the source.

TABLE 9-1D. THE HEALTH COST PER KG OF MOTOR-VEHICLE AND RELATED UPSTREAM AND ROAD-DUST EMISSIONS, BASED ON A 10% REDUCTION IN DIRECT MOTOR-VEHICLE EMISSIONS, RELATED UPSTREAM EMISSIONS, PAVED-ROAD-DUST EMISSIONS, AND UNPAVED-ROAD-DUST EMISSIONS (1990 EMISSIONS, 1991$/KG-EMITTED)

Emission	**Ambient pollutant**	**United States**		**All urban areas**		**Los Angeles**	
		low	*high*	*low*	*high*	*low*	*high*
CO	CO	0.01	0.09	0.01	0.10	0.03	0.18
NO_x	nitrate PM_{10}	0.96	15.53	1.31	21.17	6.02	75.11
	NO_2	0.14	0.68	0.18	0.91	0.52	2.62
Total for NOx		1.10	16.21	1.50	22.08	6.54	77.73
$PM_{2.5}$	$PM_{2.5}$	3.22	45.22	6.53	88.79	41.93	405.29
$PM_{2.5-10}$	$PM_{2.5-10}$	0.27	2.95	0.63	6.20	4.69	29.77
Total for PM_{10}		0.60	15.13	1.45	31.69	12.35	155.58
SO_x	sulfate PM_{10}	2.80	22.60	4.40	35.28	33.53	209.88
VOC	organic PM_{10}	0.10	0.99	0.13	1.25	0.52	4.29
VOC+NO_x	ozone	0.01	0.10	0.02	0.12	0.05	0.39

Source: Report #11. Note that this table includes emissions of dust from unpaved roads, whereas the previous table does not.

Each $/kg value is equal to the total calculated health damages attributable to the pollutant and source, divided by emissions of the pollutant from the source.

TABLE 9-2. THE VISIBILITY COST PER KG OF MOTOR-VEHICLE AND RELATED EMISSIONS, BASED ON A 10% REDUCTION IN EMISSIONS (1990 EMISSIONS, 1991$/KG-EMITTED)

Emissions source	**$/kg-$PM_{10}$[a]**		**$/kg-$NO_x$(b)**		**$/kg-$SO_x$(c)**		**$/kg-VOCs[d]**	
	Low	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*
MVs	0.40	3.90	0.19	1.11	0.89	3.97	0.01	0.05
MVs + U	0.37	3.47	0.17	1.04	0.36	1.35	0.01	0.04
MVs + U + RDP	0.32	2.07	0.17	1.04	0.36	1.35	0.01	0.04
MVs + U + RDP +RDU	0.10	0.81	0.17	1.04	0.36	1.35	0.01	0.04

Source: Report #13.

MVs = motor vehicles; U = upstream emissions related to motor-vehicle use (e.g., from petroleum refineries); RDP = road dust from paved roads; RDU = road dust from unpaved roads; PM_{10} = particulate matter of aerodynamic diameter of 10 microns or less; NO_x = nitrogen oxides; SO_x = sulfur oxides; VOCs = volatile organic compounds.

Note that in all cases, the year of the analysis is 1990 (i.e., 1990 data for emissions, air quality, and income), but the year of the dollars is 1991.

[a]Equal to the dollar cost of light extinction caused by 10% of the primary ambient PM_{10} attributable to motor vehicles, divided by 10% of PM_{10} emissions attributable to motor vehicles. Primary or direct PM is PM that is emitted as such, as distinguished from PM that is formed in the atmosphere.

[b]NO_x emissions can become ambient NO_2 or form particulate nitrate aerosols. The $/kg estimate here is equal to the dollar cost of light extinction caused by 10% of the ambient NO_2 and 10% of the ambient particulate nitrate attributable to motor vehicles, divided by 10% of NO_x emissions attributable to motor vehicles.

[c]SO_x emissions can form particulate sulfate aerosols, which scatter light and reduce visibility. The $/kg estimate here is equal to the dollar cost of light extinction caused by 10% of the ambient particulate sulfate attributable to motor vehicles, divided by 10% of SO_x emissions attributable to motor vehicles.

[d]VOC emissions can form secondary organic aerosols, which scatter light and reduce visibility. The $/kg estimate here is equal to the dollar cost of light extinction caused by 10% of the ambient organic aerosol attributed to motor vehicles, divided by 10% of VOC emissions

TABLE 9-3. THE CHANGE IN WELFARE IN THE CROP MARKET DUE TO A 10% OR 100% REDUCTION IN MOTOR-VEHICLE RELATED EMISSIONS (1990 EMISSIONS, 1991$/1000-VMT AND 1991$/KG-[NO_X+VOCS])

	$/1000-VMT				**$/kg-[VOCs+$NO_X$][a]**			
	Direct emissions[b]		**Direct + upstream[c]**		**Direct emissions[b]**		**Direct + upstream[c]**	
Case IIA: 10% reduction[d]	*Low*	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*
LDGAs	0.99	1.77	1.13	1.91	0.19	0.29	0.17	0.26
LDGTs	1.67	2.97	1.91	3.21	0.20	0.31	0.18	0.27
HDGVs	3.99	6.56	4.57	7.18	0.14	0.23	0.13	0.21
All gasoline vehicles	*1.16*	*2.08*	*1.33*	*2.24*	*0.19*	*0.29*	*0.17*	*0.26*
LDDAs	0.37	0.57	0.40	7.44	0.21	0.33	0.19	3.47
LDDTs	0.13	0.22	0.20	0.28	0.21	0.33	0.14	0.20
HDDVs	3.40	5.74	3.69	6.02	0.19	0.32	0.17	0.28
All diesel vehicles	*2.69*	*4.53*	*2.93*	*4.76*	*0.19*	*0.32*	*0.17*	*0.28*
All gasoline, diesel vehicles	*1.29*	*2.28*	*1.46*	*2.45*	*0.19*	*0.30*	*0.17*	*0.26*
Case IIB: 100% reduction[e]	**1.37**	**2.51**	**1.53**	**2.68**	**0.20**	**0.33**	**0.18**	**0.29**

Source: Report #12.

LDGA = light-duty gasoline auto; LDGT = light-duty gasoline truck; HDGV = heavy-duty gasoline vehicle; LDDA = light-duty diesel auto; LDDT = light-duty diesel truck; HDDV = heavy-duty diesel vehicle; VMT = vehicle-miles of travel.

These results include the effect of ozone on all crops, and the effects of pollutants other than ozone.

[a]Includes a minor amount of damage (5-10%) attributable to pollutants other than ozone, the pollutant formed from NO_X and VOC emissions.

[b]Direct emissions are tailpipe and evaporative emissions from motor vehicles.

[c]Upstream emissions include emissions from the production of motor fuels, the servicing of motor vehicles, the production of crude oil used to make motor fuel, the production of motor vehicles, and so on. See Report #10 for details.

[d]Case IIA is a 10% reduction in emissions of VOCs and NOx.

[e]Case IIB is a 100% reduction in emissions of VOCs and NOx.

TABLE 9-4. COST OF CLIMATE CHANGE DUE TO FUEL-CYCLE CO_2-EQUIVALENT EMISSIONS OF GREENHOUSE GASES FROM MOTOR-VEHICLES (1991 EMISSIONS, 1991 $)

	Light-duty automobiles		Light-duty trucks		Heavy-duty vehicles		**Fleet total**	
	Low	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*
Gasoline vehicles								
Annual VMT (billion)[a]	1,525	1,525	439	439	24	24	**1,987**	**1,987**
LC CO_2-equiv. (g/mi)[b]	800	860	1,040	1,100	2,150	2,250	n.e.	n.e.
Total CO_2 equivalent (10^{12} grams)[c]	1,220	1,311	457	483	51	53	**1,727**	**1,848**
Diesel vehicles								
Annual VMT (10^9 mi)[a]	18	18	13	13	154	154	**185**	**185**
LC CO_2-equiv. (g/mi)[b]	1,550	1,700	1,800	1,900	3,800	4,000	n.e.	n.e.
Total CO_2 equivalent (10^{12} grams)[c]	28	31	23	25	584	614	**635**	**670**
All vehicles								
Damage cost of CO_2 in U.S. (\$/10^6-g-CO_2)[d]	0.00	1.40	0.00	1.40	0.00	1.40	0.00	1.40
Total cost (billion $)[e]	**0.00**	**1.88**	**0.00**	**0.71**	**0.00**	**0.93**	**0.00**	**3.52**

n.e. = not estimated. **Light-duty automobiles** are passenger vehicles, including station wagons and motorcycles; **light-duty trucks** are trucks, vans, minivans, jeeps, and utility vehicles, with a gross vehicle weight rating of 8,500 lbs or less and a curb weight of 6,000 lbs or less; and **heavy-duty vehicles** are all other trucks, and buses.

[a] Annual miles of travel by vehicle class, from Table10-3 of Report #10.

[b] Lifecycle (LC) CO_2-equivalent emissions of all greenhouse gases (see equation 9-13) from the entire lifecycle of fuels and vehicles, per mile of travel. Ranges estimated based on results from the Lifecycle Emissions Model (LEM) (Delucchi, 2003). See the text for further discussion.

[c] Equal to g/mile CO_2-equivalent fuel-cycle emissions multiplied by total VMT of travel.

[d] Based on the literature review and analysis presented in the text. A million grams is a metric tonne, equal to 1.102 English tons (2000 lbs).

[e] Equal to the $/metric-tonne cost multiplied by total lifecycle CO_2-equivalent emissions for gasoline and diesel.

TABLE 9-5. SUMMARY OF ESTIMATES OF GLOBAL DAMAGE COST OF GLOBAL WARMING

	Damage %	with respect to	1991\$/ton-C	1991\$/ton-CO_2
Cline (1992) (2.5^o C short-term warming)	1.0 – 2.0	world GDP	4 – 40	1.1 –11
Cline (1992) (10^o C long-term warming)	6.0 –12.0	world GDP	220?	60?
Nordhaus (1993)	1.33	world output	5 – 22	1.4 – 5.8
Nordhaus (1991)	0.25 – 2.0	world output	0.3 – 71	0.1 – 19
Ayres & Walters (1991)	2.1 – 2.4	GWI	40 – 46	11 – 13
Fankhauser (1994)[e]	n.e.	n.e.	6 – 66	1.7– 18
Titus (1992)	n.e.	n.e.	40 - 530 (165)	11 - 143 (45)
Pearce et al. (1992)	1 – 3	GWP	9 – 28	2.5 – 7.7
Hohmeyer (1996)	n.e.	n.e.	800	220
CEC (1992)	n.e.	n.e.	30	8.2
Montgomery (1991)	0.5	GNP	n.e.	n.e.
Tol (1995)	1.9	world GDP	n.e.	n.e.
Tol (1999)	n.e.	n.e.	9 – 180	3 – 50
Goodland & El Serafy (1998)	n.e.	n.e.	23	6.2
Tol (2003a)	n.e.	n.e.	1 – 20	0.3 – 5
Tol (2003b)	-2.3 – 2.7	World GDP	~ 4 – 40	~ 1 – 10
Pearce (2003)	n.e.	n.e.	3 – 32	1 – 9

GWI = gross world income; GWP = gross world product; GNP = gross national product; GDP = gross domestic product; n.e. = not estimated; C = carbon; CO_2 = carbon dioxide; CEC = California Energy Commission. A "ton" is 2,000 lbs; a metric "tonne" is 1000 kg.

See the text for details.

TABLE 9-6. THE MARGINAL COST OF NOISE FROM A 10% INCREASE IN VMT, FOR DIFFERENT TYPES OF VEHICLES ON DIFFERENT TYPES OF ROADS, IN URBANIZED AREAS (1990 TRAVEL, 1991$/1000-VMT)

	Interstate	Other freeways	Principal arterials	Minor arterials	Collectors	Local roads
Base case						
LDAs	2.96	4.25	1.18	0.57	0.07	0.00
MDTs	8.50	13.20	7.02	5.37	1.05	0.00
HDTs	16.69	30.80	20.07	29.93	4.93	0.00
Buses	6.36	9.77	7.18	6.42	1.22	0.00
Motorcycles	17.15	27.03	8.71	4.67	0.56	0.00
Low-cost case						
LDAs	0.11	0.18	0.04	0.01	0.00	0.00
MDTs	0.40	0.66	0.32	0.18	0.01	0.00
HDTs	0.81	1.62	1.22	1.77	0.06	0.00
Buses	0.35	0.58	0.38	0.22	0.00	0.00
Motorcycles	0.66	1.13	0.27	0.09	0.00	0.00
High-cost case						
LDAs	40.11	56.02	16.20	9.35	6.04	0.44
MDTs	114.76	173.38	96.05	84.93	78.84	12.13
HDTs	225.61	404.82	269.27	414.17	319.22	92.04
Buses	86.15	128.60	98.66	105.33	108.00	12.84
Motorcycles	232.47	355.73	119.64	76.65	50.08	2.73

Source: Report #14.

VMT = vehicle miles of travel; LDAs = light-duty autos; MDTs = medium-duty trucks; HDTs = heavy-duty trucks.

TABLE 9-7. NONMONETARY COSTS OF MOTOR-VEHICLE RELATED CRIMES

Crime category	VCT [a]	Unit costs ($/VCT) [b]				SUBF [c]		Total cost [d] (10^6 $)	
		Lost productivity		Pain and suffering					
		Low	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*
Murder of police during traffic stop	13	50,000	200,000	637,610	2,550,440	0.30	0.70	2.7	25.0
Murder during motor-vehicle theft	81	50,000	200,000	637,610	2,550,440	0.50	0.90	27.8	199.8
Murder during robbery of gas station	89	50,000	200,000	637,610	2,550,440	0.25	0.50	15.3	122.2
Rape or sexual assault in a parking lot or garage*	48,846	213	427	56,901	94,834	0.10	0.25	279.0	1,163.3
Robbery of gas station	17,829	100	114	3,793	5,216	0.20	0.50	13.9	47.5
Robbery in parking lot or garage*	136,255	100	114	3,793	5,216	0.05	0.20	26.5	145.2
Robbery of MVs ("car-jackings")*	37,000	100	114	3,793	5,216	0.50	0.80	72.0	157.8
Theft of autos and motorcycles*	1,742,672	6	7	190	379	0.50	0.80	170.2	538.8
Theft of trucks and buses*	310,513	6	7	190	379	0.50	0.80	30.3	96.0
Larceny theft from MVs*	2,810,533	1	1	50	100	0.20	0.50	28.6	142.5
Larceny theft of MV accessories*	1,769,272	1	1	50	100	0.50	0.80	45.0	143.6
Other traffic violations	458,167	2	5	0	0	1.00	1.00	0.9	2.3
Fraud, receiving stolen property, others	30,590	2	5	50	100	0.50	0.90	0.8	2.9

VCT = victimizations; SUBF = the fraction of crimes for which there would have been no substitute; MV = motor vehicle. A victimization is "a crime as it affects one individual person or household. For personal crimes, the number of victimizations is equal to the number of victims involved. The number of victimizations may be greater than the number of incidents because more than one person may be victimized during an incident. Each crime against a household is assumed to involve a single victim, the affected household"(BJS, 1992, p. 156).

* Includes attempted crimes.

[a]*Murder:* I presume that the number of victims is equal to the number reported to the police (Table 7-10, Report #7).

Rape: Miller et al. (1995) estimate that there were an average of 1.163 million attempted or completed rapes and sexual assaults (excluding child abuse, and taking a conservative view of "series" victimizations) per year from 1987 to 1990. This figure substantially exceeds the number of rapes reported by the NCVS (e.g., BJS, 1992), primarily because until recently the NCVS asked not about rape specifically but rather about "attacks" in general (BJS, 1992), and in part because non-rape sexual assault was not included (Miller et al., 1995). Of the rapes actually reported to the [old] NCVS in 1991, 4.2% occurred in a parking lot or garage (BJS, 1992). I assume therefore that 4.2% of the 1.163 million estimated by Miller et al. (1995) occurred in a parking lot or garage.

Robbery of gas station: I assume that the number of victimizations is equal to the number of offenses known to the police (Table 7-10, Report #7), because it seems probable that virtually all gas station robberies are reported to the police.

Robbery in parking lot: In 1991 there were 1.145 million victims of attempted or completed robbery (whereby the same person victimized twice counts as two victims), 11.9% of which occurred in a parking lot or garage (BJS, 1992).

Carjacking: According to *National Crime Victimization Survey* (NCVS), there were 35,500 attempted or completed carjackings per year from 1987 to 1992 (BJS, 1994). It is likely that the number increased from 1987 to 1992, so that the number in 1991 was greater than this five-year annual average.

MV theft: Victims reported 2.11233 million completed or attempted motor-vehicle thefts for 1991 (BJS, 1992). I assume that these were distributed to vehicle types in the same proportion as were thefts reported to the police (Report #7).

Larceny theft: Victims reported 12.52163 million completed or attempted larceny thefts for 1991 (BJS, 1992). I assume that thefts from motor vehicles, or of motor-vehicle accessories, were the same fraction of this the total as they were of larceny thefts reported to the police (Report #7).

Other traffic violations, fraud etc.: I assume the number of arrests, from Table 7-10 (Report #7).

[b]*Murder:* The pain and suffering cost is equal to VOL·PSF, and the non-monetary lost-productivity cost is equal to VOL·NMLP, where VOL is our estimate of the total value of a life ($1,000,000 to $4,000,000; Report #11), NM is the fraction of the total value that is due to pain and suffering (0.64; based on Miller [1997] and Blincoe [1996]), and NMLP is the fraction of the total that is due to lost nonmonetary productivity (0.05, based on Miller [1997] and Blincoe [1996]).

Rape: The pain and suffering cost is based on the estimate of Miller et al. (1995), shown below. I bracket the Miller et al. estimate with low and high values, then multiply by the by the 1993/1991 GNP implicit price deflator (0.948), because the Miller et al. estimates are in 1993 dollars. The non-monetary lost productivity cost is equal to the Miller et al. (1995) estimate of the total monetary+nonmonetary lost productivity cost, multiplied by the nonmonetary cost fraction of the total and by the 1993/1991 GNP implicit price deflator. On the basis of data in Blincoe (1996), I estimate that non-monetary lost productivity is 15% of total lost productivity, which includes wages, fringe benefits, housework, school days, job training, insurance administration, and legal expenses.

Robbery: The method is the same as for *rape*, above, except that I first calculate from the Miller et al. (1995) estimates an injury-weighted average robbery cost, equal to RWI·RPIF+RWOI·(1-RPIF), where RWI is the cost of robbery with injury, RPIF is the fraction of parking-lot or garage robberies that results in an injury (0.275; BJS, 1992), and RWOI is the cost of a robbery without injury.

Motor-vehicle theft: The method is the same as for *rape*.

Larceny theft: The pain and suffering cost is my estimate. The non-monetary lost-productivity cost is estimated as for *rape.*

Other traffic violations, fraud etc.: my assumptions.

These are the Miller et al. (1995) estimates (1993 $):

Crime category	*pain & suffering (PS)*	*lost productivity (LP)*
Rape and sexual assault	81,400	2,200
Robbery or attempt, with injury	13,800	2,500
Robbery or attempt, without injury	1,300	75
Larceny theft or attempt	0	8
Motor-vehicle theft or attempt	300	45

[c]See Table 7-10 and discussion thereof in Report #7.

[d]See equation 9-19.

[e]As noted in the text, I do not include in my final totals the cost of murders of police officers, because the corresponding costs presumably are reflected in the compensation of officers.

TABLE 9-8. SUMMARY OF THE COST PER KG OF MOTOR-VEHICLE AND RELATED EMISSIONS IN URBAN AREAS, BASED ON A 10% REDUCTION IN EMISSIONS (1990 EMISSIONS, 1991$/KG-EMITTED)

Emitted --> / *Ambient -->*	**PM_{10}** / *PM_{10}*		**NO_x** / *NO_2, nitrate PM_{10}*		**SO_x** / *sulfate PM_{10}*		**CO** / *CO*		**VOCs** / *organic PM_{10}*		**VOCs + NO_x** / *O_3*	
	Low	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*
Health												
MVs	13.7	188	1.6	23.3	9.6	90.9	0.0	0.1	0.1	1.5	0.0	0.1
MVs + U	12.2	158	1.5	22.1	4.4	35.3	0.0	0.1	0.1	1.3	0.0	0.1
MVs + U + RD	1.5	31.7	1.5	22.1	4.4	35.3	0.0	0.1	0.1	1.3	0.0	0.1
Visibility												
MVs	0.4	3.9	0.2	1.1	0.9	4.0	0.0	0.0	0.0	0.1	0.0	0.0
MVs + U	0.4	3.5	0.2	1.0	0.4	1.3	0.0	0.0	0.0	0.1	0.0	0.0
MVs + U + RD	0.1	0.8	0.2	1.0	0.4	1.3	0.0	0.0	0.0	0.1	0.0	0.0
Crops												
MVs	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.	0.0	0.0	0.0	0.0	0.2	0.3
MVs + U	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.	0.0	0.0	0.0	0.0	0.2	0.3
MVs + U + RD	n.e.	n.e.	n.e.	n.e.	n.e.	n.e.	0.0	0.0	0.0	0.0	0.2	0.3
Climate change[a]	0.0	2100	0.0	5.6	0.0	-56	0.0	16.8	0.0	n.e.	n.e.	n.e.

[a] $/Mg. Equal to the damage cost in the U. S. per kg of CO_2 (Table 9-4) multiplied by the CO_2 equivalency factor for each pollutant (Appendix D of Delucchi [2003]). Global damages would be at least 7 times higher.

Source: Tables 9-1a through Table 9-4.

TABLE 9-9. SUMMARY OF COST ESTIMATES

A. THE NONMONETARY EXTERNAL COSTS OF MOTOR-VEHICLE USE, 1990-91 (10^9 1991$)

Cost item	*Low*	*High*	*Q[a].*
Accidental pain, suffering, death, and lost nonmarket productivity not accounted for by economically responsible party	9.5	97.7	A2/B
Travel delay, imposed by others, that displaces unpaid activities	22.5	99.3	A2
Air pollution: human mortality and morbidity due to particulate[b] emissions from vehicles	16.7	266.4	A1
Air pollution: human mortality and morbidity due to all other pollutants from vehicles	2.3	17.1	A1
Air pollution: human mortality and morbidity, due to all pollutants from upstream processes	2.3	13.0	A1
Air pollution: human mortality and morbidity from road dust	3.0	153.5	A1
Air pollution: loss of visibility, due to all pollutants attributable to motor vehicles	3.6	27.4	A1
Air pollution: damage to agricultural crops, due to ozone attributable to motor vehicles	3.3	5.7	A1
Air pollution: damages to materials, due to all pollutants attributable to motor vehicles	1.0	8.0	B [A1]
Air pollution: damage to forests, due to all pollutants attributable to motor vehicles	0.2	2.0	B [A2]
Climate change due to lifecycle emissions of greenhouse gases (U. S. damages only)	0.0	3.5	A1, B [A1][c]
Noise from motor vehicles	0.5	15.0	A1
Water pollution: health and environmental effects of leaking motor-fuel storage tanks	0.1	0.5	D
Water pollution: environmental and economic impacts of large oil spills	0.2	0.5	C [A1]
Water pollution: urban runoff polluted by oil from motor vehicles	0.1	0.5	D[d]
Nonmonetary costs of net crimes related to using or having motor-vehicle goods, services, or infrastructure	0.7	2.8	A3
Nonmonetary costs of fires related to using or having motor-vehicle goods, services, or infrastructure	0.0	0.2	A3
Total	**65.6**	**714.7**	

B. NONMONETARY EXTERNAL COSTS OF MOTOR-VEHICLE USE NOT ESTIMATED IN THIS REPORT

Cost item	*Low*	*High*	*Q[a]*
Air pollution: damages to natural ecosystems other than forests, due to all pollutants attributable to motor vehicles	n.e.	n.e.	n.a.
Water pollution: health and environmental effects of leaking solid-waste storage sites	n.e.	n.e.	n.a.
Vibration damages from motor vehicles	n.e.	n.e.	n.a.
Fear and avoidance of motor vehicles and crimes related to motor-vehicle use	n.e.	n.e.	n.a.

C. NONMONETARY ENVIRONMENTAL AND SOCIAL COSTS OF THE MOTOR-VEHICLE INFRASTRUCTURE

Cost item	*Low*	*High*	*Q[a]*
Land-use damage: habitat destruction and species loss due to highway and motor-vehicle infrastructure	n.e.	n.e.	n.a.
Water pollution: urban runoff polluted by oil from motor vehicles, and pollution from highway deicing	0.6	1.2	D
The socially divisive effect of roads as physical barriers in communities	n.e.	n.e.	n.a.
The esthetics of highways and service establishments	n.e.	n.e.	n.a.
Total	**n.e.**	**n.e.**	

n.e. = not estimated; n.a. = not applicable.

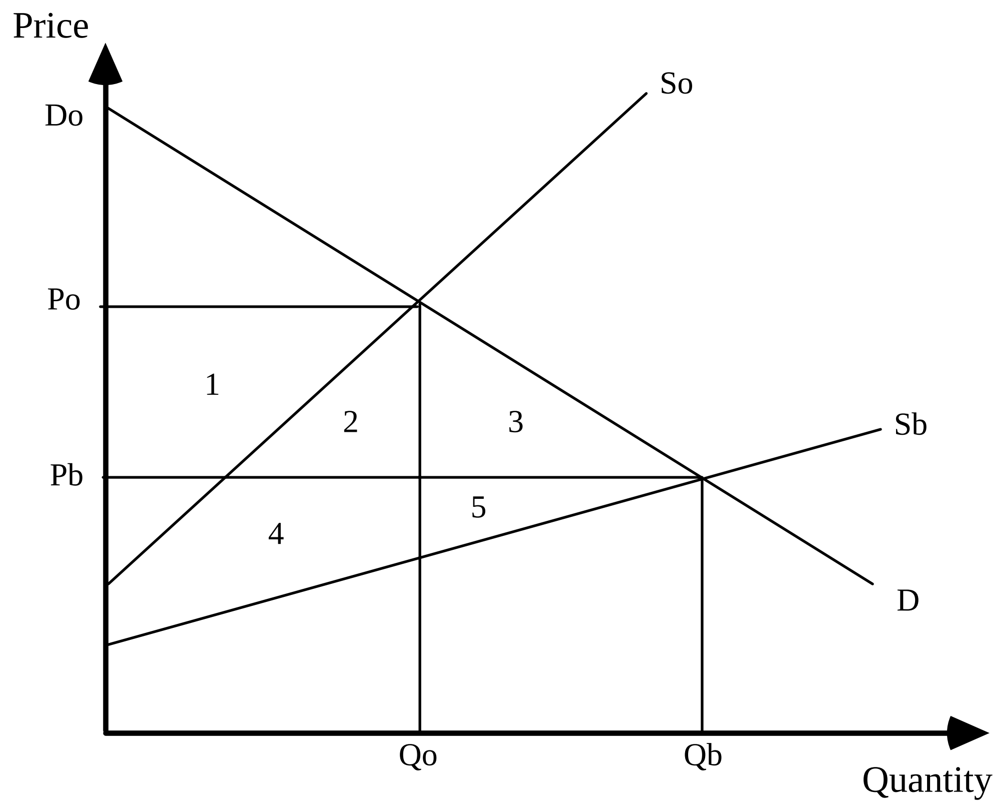
[a]Q = Quality of the estimate (see Table 1-3 of Report #1). Ratings in brackets refer to the quality of the analysis in the literature reviewed.

[b]Includes secondary PM, formed from direct emissions of SO_x, NO_x, and NH_3.

[c]The estimate of lifecycle emissions of greenhouse-gases is original and detailed (A1), whereas the estimate of the $/ton cost of emissions is based on a review of literature (B) that reports the results from detailed model calculations ([A1]).

[d] This is my estimate of the cost as of 1997. As discussed in the text, the cost probably was higher in 1991, because the leakage-prevention and clean-up programs were not in place everywhere. I speculate that the external costs in 1991 were three times the costs today.

FIGURE 9-1. THE EFFECT OF AIR POLLUTION ON AGRICULTURAL PRODUCTION



See section 9.2.5 for discussion.

The superscript ***o*** refers to current (1990) ozone levels, and the superscript ***b*** refers to background ozone level or the level without motor-vehicle related pollution.

This diagram is for illustrative purposes only. No inferences should be made about the relative sizes of the various regions shown in this diagram.