Environ Resource Econ (2018) 69:591–608
https://doi.org/10.1007/s10640-017-0214-8

Can Understanding Spatial Equilibria Enhance Benefit Transfers for Environmental Policy Evaluation?

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Accepted: 1 December 2017 / Published online: 2 January 2018

Abstract A conceptual model of consumer sorting in markets for housing, labor and health care is outlined and used to make three points about how benefit transfers are used for environmental policy evaluation. First, the standard approach to assessing benefits of air quality improvements by transferring the value of a statistical life from labor market studies embeds several untested (but testable) assumptions. Second, if the cost of an environmental policy exceeds its capitalized effect on housing prices, then the capitalization effect is an insufficient statistic for determining whether benefits exceed costs. Third, there are several ways in which equilibrium sorting models may be usefully extended to assess distributional welfare effects of environmental policies.

1 Introduction

When it comes to environmental amenities, the choice of where to live is probably the single most important choice that most people ever make. A person's residential location influences their exposures to air pollution, water pollution, extreme weather and hazardous waste sites. People differ in their preferences for these amenities, in their resilience to pollution, and in how much they are willing to pay to improve features of environmental quality. Tiebout (1956) notion that people can effectively purchase their preferred amenity bundles by "voting

Guest Editor: V. Kerry Smith.

I am grateful to Jared Carbone and V. Kerry Smith for helpful comments and suggestions on this paper. I also benefitted from discussions with Kelly Bishop, Jonathan Ketcham, Alvin Murphy, Sophie Mathes, Nirman Saha and participants in the December 2016 EPA Workshop on "Benefit Transfer: Evaluating How Close is Close Enough".

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with their feet" has formed the basis for equilibrium sorting models and hedonic property values models that have long served as a workhorse of non-market valuation. Yet federal benefit-cost analyses of environmental policies affecting human health do not attempt to fully utilize information about consumers' willingness to pay that is potentially revealed by spatial equilibrium in housing markets. Instead, they rely primarily on estimates for the value of a statistical life (VSL) transferred from labor market studies. This paper asks if the predominant focus on VSL transfers results in leaving useful information on the table that could potentially improve benefit-cost analysis of environmental policies.

I outline a conceptual framework for linking the primary ways in which environmental regulations affect consumer welfare to the primary ways in which those welfare effects may be revealed by the choices consumers make in markets for housing, labor and health care. The framework extends Bieri et al. (2014) static model of spatial equilibrium in the housing and labor markets to include a health care sector. I use the framework to highlight some of the key economic assumptions underlying the way that federal agencies estimate the monetary benefits of prospective policies by transferring VSL estimates between different contexts, time periods, and subpopulations. I also use the framework to explain how retrospective studies estimating the housing market capitalization effects of past policies can help to inform ex post judgments about their efficacy. I conclude by suggesting ways to extend the current generation of equilibrium sorting models to develop a framework better suited to evaluating distributional welfare implications of national environmental policies. The goal of such a framework would be to enable researchers and policy analysts to replace benefit transfers with a more precise and detailed approach to policy evaluation.

2 Background

To motivate the rest of the paper, I begin with an example of how spatial equilibrium models can imply substantially different estimates for the benefits of environmental regulations compared to federal benefit transfer calculations. Consider air quality. The US Environmental Protection Agency (EPA) has conducted detailed analyses of the benefits and costs of its regulation on air pollutants, and numerous academic studies have sought to estimate components of those benefits and costs using a wide variety of methods. The richness of this literature makes air pollution a convenient example, but the issues raised apply more broadly. Section 5 emphasizes this by discussing implications for evaluating the benefits and costs of regulations on water pollution and hazardous waste.

Figure 1 reports several estimates for the annual benefits and costs of air quality improvements induced by the Clean Air Act Amendments (CAAA) in 2020. All numbers are reported in constant year 2006 dollars. The first column shows EPA's (2011) \$65 billion estimate for the regulatory costs imposed on firms. This statistic is based on a full employment model that excludes the regulations' effects on workers. Walker (2013) estimates that the new constraints on regulated firms cost workers approximately \$7.7 billion over a seven year period, partly due to unemployment spells and partly due to lower earnings when the workers eventually re-entered the labor force. An annualized version of this cost is included in the figure's second column.[1] The middle two columns show estimates for benefits that I calculated based on results reported by Chay and Greenstone (2005) and Bayer et al. (2009). Both studies use properties of spatial equilibria in quasi-national studies of U.S. housing markets to estimate

[1] This yields a conservatively high \$1.09 billion estimate for the annualized cost. Walker shows that the regulation's effect on earning is indistinguishable from zero after 7 years.

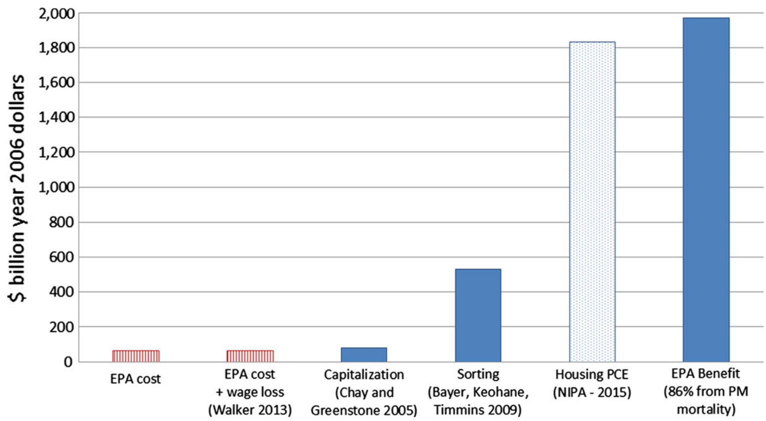

Fig. 1 Estimated benefits and costs of CAAA reductions in air pollution in 2020

households' average willingness to pay for a marginal improvement in air quality (MWTP).[2] The bars in the figure are based on multiplying their main estimates for MWTP by the population-weighted average reduction in particulates smaller than 2.5 microns ($PM_{2.5}$) that EPA attributes to the CAAA projected into the year 2020 (9 micrograms per cubic meter).[3] Further, since the calculations treat demand as perfectly elastic, they are best interpreted as upper bounds on partial equilibrium benefits for the relevant populations. As a reference point, the second to last column shows that total US housing expenditures in 2015 were approximately $1.8 trillion based on the National Income and Product Accounts. Finally, the last column shows EPA's (2011) benefit estimate, of which 86% comes from multiplying an estimate for the number of lives saved from reduced ambient concentrations of $PM_{2.5}$ by a composite $8.8 million VSL estimate transferred from a series of studies, most of which focus on wage-risk tradeoffs in the labor market.

Comparing the two cost measures with the three benefit measures shows that the choice among these different approaches to measuring benefits does not alter EPA's conclusion that the benefits of CAAA outweigh the costs.[4] However, the choice among methods has a large

[2] I refer to these studies as quasi-national because they use data from a large but incomplete portion of the United States, focusing on nonrandom subsets of the people who live in the areas they study. Specifically, Chay and Greenstone's estimates are based on the median self-assessed value of owner occupied houses in approximately one third of US counties. Bayer, Keohane and Timmins focus on household heads under the age of 35 living in 242 metropolitan statistical areas.

[3] Specifically, I start with Chay and Greenstone's (2005) household MWTP estimate of $243 per permanent microgram per cubic meter reduction of total suspended particulates (TSP) and Bayer's et al. (2009) household estimate of $149 in annualized MWTP per microgram per cubic meter reduction in PM_{10}. Then I convert TSP and PM_{10} to $PM_{2.5}$ using a conversion factor of 1.82 to go from TSP to PM_{10} as suggested by Bayer, Keohane and Timmins and a conversion factor of 0.55 to go from PM_{10} to $PM_{2.5}$ based on my calculations from a population-weighted regression of $PM_{2.5}$on PM_{10} in the year 2000. Since Chay and Greenstone measure MWTP for a permanent reduction in air pollution, I annualize their measure assuming a user cost of housing of 7.86% based on Blomquist et al. (1988). Finally, I assume 123 million households in 2020 based on multiplying the Census Bureau's 1-year ACS estimate for 2015 (118 million) by their projection for U.S. population growth between 2015 and 2020 (4.1%). All dollar values are converted to year 2006 dollars using the GDP deflator.

[4] The capitalization based measure derived using Chay and Greenstone (2005) estimate is $79 billion compared to $66 billion in total costs defined by adding Walker's (2013) estimate for labor market costs to EPA's cost estimate.

effect on the level of net benefits.[5] This suggests that the method used could determine the sign of net benefits in other benefit-cost analyses. With this in mind, Fig. 1 raises at least three questions about the way in which estimates for the benefits of air quality improvements and mortality reductions are transferred to assess the benefits of federal environmental regulations. First, why does Bayer et al. (2009) model of household sorting behavior yield benefit estimates that are approximately seven times as large as those based on Chay and Greenstone's (2005) estimate for how air quality improvements are capitalized into housing prices? Second, if the willingness to pay for air quality is capitalized into housing prices, then how can EPA's benefit estimates exceed total annual housing expenditures in 2015? Third, does an \$8.8 million VSL make sense for valuing $PM_{2.5}$mortality? To help answer these questions, I begin by sketching a simple conceptual model of spatial equilibrium in the housing, labor and health care sectors.

3 A Model of Spatial Equilibrium: Housing, Labor and Health Care

Consider a static sorting model of the United States in which firms choose locations and hire workers to maximize profit and households choose occupations and locations to maximize utility. The country can be divided into $j = 1, \ldots, J$ distinct locations such as counties or metropolitan areas that differ in the wages paid to workers, w_j, in the annualized after-tax price of land, r_j, and in a vector of K environmental amenities, $g_j = \left[g_{1j}, \ldots, g_{Kj}\right]$. Examples include features of climate, ambient air quality and water quality, and proximity to recreation areas and hazardous waste sites.[6] At each location workers may engage in $k = 1, \ldots, K$ occupations, where the set of potential occupations is defined broadly to include $K - 2$ types of paid labor, retirement and unemployment.

3.1 Annual Mortality Risks

Individuals face baseline age-specific mortality rates that are independent of their occupations and spatial locations, d_a.[7] Conditional on age, each individual faces an additional risk of death, d_{aj}, from exposure to environmental externalities in their neighborhoods. Examples include fatal heart attacks and strokes triggered by spikes in air pollution or temperature. Individuals also face mortality risks specific to their occupations, d_{ak}. For workers, the risk would stem from fatal on-the-job accidents. For retirees and unemployed workers the "occupational" risk would stem from longer duration of exposures to ambient pollutants in their neighborhoods. Hence the total annual mortality risk for an age-a individual living in neighborhood $j$and working in occupation k is $d_{ajk} = d_a + d_{aj} + d_{ak}$.

People can adjust each component of their annual mortality risk rate through a separate market mechanism. Workers can reduce d_{ak} by moving to jobs with safer working conditions. Households can reduce d_{aj} by moving to neighborhoods with cleaner air and milder climates. These migration-based risk reductions may come at a cost of lower wages and higher rents.

[5] Sullivan (2017) makes a similar observation and suggests that at least part of the discrepancy is because the standard approaches that economists use to assign air pollution exposures to people at their residential locations introduce severe measurement error that attenuates hedonic estimates of their marginal willingness to pay for air quality.

[6] I abstract from other local public goods and non-environmental amenities that affect the quality of life in Roback (1982) style models to avoid extraneous notation and to focus attention on environmental policy. Other amenities could be added to without altering the main points of this paper.

[7] One might prefer to condition on a broader set of characteristics when defining baseline mortality risk such as gender, race and genetic markers. Here I condition on age alone for notational simplicity and to help relate the conceptual framework to VSL estimates from Hall and Jones (2007).

Similarly, individuals can pay to reduce d_a by investing in health care.[8] Let p_{aj} represent the cost to an age-a individual of reducing their baseline mortality risk by one unit (e.g. 1 chance of death in 100,000).[9] Such reductions might be achieved through medical procedures, drugs, investments in diet and exercise, or preventative care. The j subscript on p_{aj} recognizes that the age-specific cost of reducing mortality risk varies across space due to heterogeneity in doctors' skills, hospital organization, and the extent to which states contribute to Medicaid and invest in other public health programs (Finkelstein et al. 2016).

3.2 Utility Maximization

Utility maximization can be envisioned as a two-stage process in which a household first determines its optimal occupation and consumption bundle at every possible spatial location and then chooses the optimal location. Equation (1) combines both stages into a single constrained optimization problem.

$$\max_{x,l,j,k} U\left(x, l, g_j, d_a, d_{aj}, d_{ak}, \alpha\right) : nw + w_{jk} = x + r_j l + p_{aj} + mc_{\alpha jk}. \tag{1}$$

Households are heterogeneous. In addition to differing in age, they differ in their job skills, preferences for amenities, and in the financial costs they face to move between neighborhoods or occupations. For notational simplicity, the parameter α is used to index all forms of heterogeneity in the sense that each α-type household has a unique combination of preferences, skills, age and moving costs. Households enjoy the quality of life provided by the bundle of amenities in their chosen locations. Each amenity may affect utility directly (e.g. certain manufacturing activities may generate noise, traffic and air and water pollution) in addition to entering utility through the effects of pollution on neighborhood mortality rates.

Notice that the specification in (1) differs from the usual expected utility maximization framework by allowing mortality risk to enter utility directly. One reason for adopting this unconventional formulation is to highlight an assumption that is embedded in the way that VSL estimates are regularly transferred between different contexts. Households may differ in their relative preferences for avoiding neighborhood-specific mortality risk relative to age-specific risk or occupation-specific risk. This is reflected in the way that d_a, d_{aj} and d_{ak} each enter the utility function (1) as separate arguments. For example, people may be willing to pay more to avoid fatal lung cancer from air pollution than they are willing to pay to avoid instantaneous death on the job from a car crash. By contrast, the standard formulation of expected utility implicitly assumes that people are indifferent to the nature of the risk; i.e. that people perceive different mortality risks to be perfect substitutes. Equation (1) nests this assumption as a special case. The second reason for putting mortality risk directly into the utility function is to simplify derivation of the first-order conditions below. Throughout, I assume that the marginal utility of mortality risk is globally negative and the marginal utilities of amenities are globally non-negative: $U_A \geq 0$ and $U_d < 0$.

Each working household supplies one unit of labor, for which it is paid according to its skills. A portion of this income is used to rent land, l, and the remainder is spent on health care and a nationally traded private good, x.[10] Thus, households maximize utility by selecting a residential location, an occupation at that location, and using their income to purchase l, h and

[8] For simplicity, I assume that these investments do not affect occupational or neighborhood mortality risk. However, it would be interesting to consider potential interactions. For example, people who take statins to address hypertension may face a lower risk of having an air-pollution induced heart attack.

[9] Hall and Jones (2007) give an example of how to estimate the age specific cost of reducing mortality risk.

[10] This composite numeraire good includes the physical characteristics of housing.

x. In the budget constraint, total income equals the sum of wages and exogenous non-wage income, nw. Households also face differentiated financial costs of moving between occupations and/or neighborhoods based, in part, on their current locations and human capital. Moving costs are represented in the budget constraint by $mc_{\alpha jk}$. The heterogeneous psychological cost of moving away from family, friends and a familiar neighborhood is reflected in the household's α-type.

3.3 Profit Maximization

Price-taking firms maximize profits from production of health care or the numeraire good by choosing spatial locations, hiring workers in each occupation, $n = \left[n_1, \ldots, n_{K-2}\right]$, investing in on-the-job safety, $s = \left[s_1, \ldots, s_{K-2}\right]$, and choosing production quantities. Let X represent the quantity of the numeraire produced by a firm. The total cost of producing X at location j depends on the firm's choices for hired labor and capital, conditional on equilibrium wages and the stringency of regulation at location j. With this in mind, the firm's profit maximization problem can be expressed as

$$\max_{j,n,s,X} \Pi = X - C_j \left(X, s, n, w_j, r_j, g_j, \beta\right), \tag{2}$$

where β indexes all sources of firm-specific heterogeneity including production technology and capital endowments other than land and investments in job safety. Job safety affects the cost of doing business because it is assumed to require costly investments, $C_s \geq 0$, and workers must be compensated to undertake riskier working conditions since $U_d < 0$. Environmental amenities at location j may also affect the cost of doing business. An example would be a firm with a dirty production technology facing stricter regulations if it locates in a county that violates federal air quality standards. Firms in the health care sector are assumed to face a profit maximization problem analogous to (2).

3.4 Equilibrium

Equilibrium occurs when rents, wages, amenities, mortality risk, the price of health care, and location choices are defined such that all markets clear and no agent would be better off by moving. Assuming that each location provides a unique combination of amenities and neighborhood mortality risk, we can use hedonic price and wage functions to describe how rents and wages vary across space in relation to amenities in a spatial equilibrium, as shown in (3)–(4).[11] Equation (5) describes how the cost of reducing mortality via health care varies across spatial locations as a function of amenities and mortality risk.[12] Such variation may arise in equilibrium from spatial sorting by health care workers with heterogeneous skills and/or spatial variation in regulation and subsidization of the health care sector.

$$r_j = r \left\{g_j, d_j; \gamma \left[F(g), G(\alpha), H(\beta)\right]\right\} \tag{3}$$

$$w_j = w \left\{g_j, d_j; \theta \left[F(g), G(\alpha), H(\beta)\right]\right\}. \tag{4}$$

$$p_{aj} = p \left\{g_j, d_j; \delta \left[F(g), G(\alpha), H(\beta)\right]\right\}. \tag{5}$$

[11] If two locations provided identical bundles of amenities and mortality risk, then additional assumptions would be needed to rule out the possibility that they sell at different prices. For example, one could guarantee existence of a price function by adding the assumptions of free mobility and full information as in Bajari and Benkard (2005).

[12] While Eq. (5) describes an equilibrium relationship it is not a traditional "hedonic" price function in the sense that amenities and mortality risk are not direct attributes of the product being purchased.

The reduced form parameters describing the shapes of the hedonic rent, hedonic wage, and health care functions (γ, θ, δ) are themselves functions of the population distributions of amenities, households, and firms denoted by F, G, and H.

In equilibrium, spatial variation in rents, wages, and the price of health care defines the implicit price of amenity consumption. All else constant, there are three ways to induce a household to move to a more polluted location: higher wages, lower rents or cheaper health care. Likewise, people can pay to lower their mortality risk by accepting a lower wage at a safer job; they can pay to live in a safer neighborhood; or they can increase their health care expenditures. Households' choices will reflect their preferences and analysts can infer those preferences using data on households' choices along with an assumption for the parametric form of utility (1). Alternatively, analysts can infer certain marginal rates of substitution by selecting parametric forms for the equilibrium equations (3)–(5) and assuming that agents are fully informed and freely mobile.

3.5 The Benefit of a Marginal Change in Mortality Risk

The model incorporates as special cases three distinct approaches that researchers have used to measure the aggregate benefit to society of a marginal reduction in a particular mortality risk rate; i.e. the VSL. Suppose we assume that households are free to choose continuous levels of each risk, moving costs are zero and that all households are fully informed. Then VSL can be calculated by first defining an individual's willingness to pay for a marginal reduction in one of the three types of mortality risk and then aggregating over the affected populations. Equations (6)–(7) depict the equilibrium conditions.

$$(\partial U/\partial d_{ak}) \,/\, (\partial U/\partial x) = \partial w/\partial d_{ak}. \quad (6)$$

$$\left(\partial U/\partial d_{aj}\right) \,/\, (\partial U/\partial x) = -\partial r/\partial d_{aj}. \quad (7)$$

$$(\partial U/\partial d_{a}) \,/\, (\partial U/\partial x) = -p_{aj}. \quad (8)$$

Labor market studies such as Kneisner et al. (2012) estimate VSL using the wage-risk tradeoff depicted in (6); hedonic property value studies such as Davis (2004) aim to estimate the rent-risk tradeoff in (7); and Hall and Jones (2007) use a dynamic version of the health production function approach in (8).

In the special case where individuals are assumed to perceive different sources of mortality risk as perfect substitutes and are also assumed to be free to make market choices that allow them to continuously adjust the level of each risk, they will optimally choose risk levels that equate their marginal costs of reducing mortality across the three markets:

$$\partial w/\partial d_{ak} = -\partial r/\partial d_{aj} = -p_{aj}. \quad (9)$$

In this case, we could estimate VSL by using population-level information on any one of the three margins and transfer the result to a potentially different margin that is relevant for policy. Indeed, this logic is implicit in EPA's approach to transferring mortality risk estimates from the labor market to evaluate the benefits of reducing air-pollution induced deaths in their $1st$ and $2nd$ Prospective Analyses of the Clean Air Act Amendments (1999; 2011).

It is important to reiterate the restrictions on the conceptual model that enable measurement of individual willingness to pay for a marginal reduction in mortality risk (MWTP) in (6)–(8) and transfer of MWTP across contexts in (9). The key assumptions include (i) full information, (ii) free mobility, (iii) the ability to choose continuous quantities of each type of morality risk and each amenity, and (iv) and identical valuation of different types of mortality risks. I discuss the implications of these assumptions for evaluating environmental policies targeting human mortality in Sect. 4.

3.6 The Benefit of an Environmental Policy

Now consider an environmental policy that shocks the spatial distribution of an environmental amenity. Such a policy may have a variety of direct and indirect effects on consumer welfare. Equation (10) defines the willingness to pay (WTP) for the change by a household who moves from location, occupation j, k in period 0 before the policy to y, z in period 1 after the policy.

$$V\left(g_j^0, d_a^0, d_{aj}^0, d_{ak}^0, w_{jk}^0 - WTP, r_j^0, p_{aj}^0; \alpha, mc_{\alpha jk}\right)$$
$$= V\left(g_y^1, d_a^1, d_{ay}^1, d_{az}^1, w_{jz}^1, r_y^1, p_{az}^1; \alpha, mc_{\alpha jk}\right). \tag{10}$$

Consumer welfare depends on the change in the amenity experienced by the household as well as the change in mortality risk, taking into account the policy's effect on mortality as well as any changes in equilibrium prices and the household's choices of occupation, residential location and medical expenditures. With multiple margins of adjustment, the benefits to households may exceed the change in housing prices. For example, some of the benefit may be reflected through changes in wages and expenditures on medical care.

The welfare measure defined in (10) abstracts from several potentially important issues like forward looking behavior, risk aversion, information search costs, and the possibility that investments in health care reduce neighborhood-specific or occupation-specific mortality. Even with these simplifying assumptions, estimating WTP is challenging in the sense that it requires modeling how consumers and firms adjust to the policy and how those adjustments feed back into prices, risk rates and amenities as markets transition to a new equilibrium. Rather than attempt to calculate this type of "general equilibrium" WTP measure directly, the EPA and academic studies typically develop strategies to approximate it. EPA's approach, exemplified by their prospective analyses of the Clean Air Act Amendments has been to decompose the problem into the sum of its parts. They approximate the WTP measure in (10) by summing a series of estimates for the WTP for hypothetical ceteris paribus adjustments to the policy that are transferred from the results of prior academic studies. For example, in EPA (1999) and (2011) aggregate WTP for large air quality improvements is approximated by adding together a series of independent estimates of the WTP for reductions in mortality and morbidities, along with the WTP for improvements in visibility and crop yields, among other effects. Meanwhile, academic studies often take a retrospective approach that focuses on identifying how prior exogenous shocks to the amenity of interest caused housing prices and rents to change, assuming that such price adjustments will approximately reveal WTP. The following two sections discuss prior evidence on the accuracy of such approximations and suggest areas for future research.

4 Are VSL Transfers Valid?

When EPA approximates the benefits of improved air quality by adding up several individual benefits, the result that they obtain is driven by the estimated value of mortality reductions. As Fig. 1 notes, 86% of EPA's \$2 trillion annual estimate for the benefits of the Clean Air Act Amendments comes from multiplying an estimate for the VSL by an estimate for the

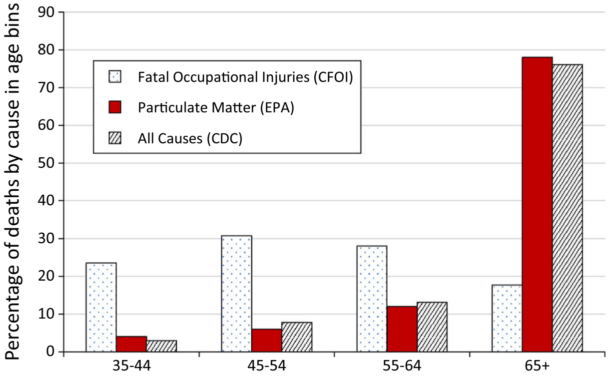

Fig. 2 Fraction of U.S. deaths by cause and age, 2010

number of lives saved. The importance of the VSL for federal policy extends far beyond air pollution and the EPA. Lee and Taylor (2014) note that up to 70% of estimated benefits in benefit-cost analyses for all federal regulations are due to mortality reductions valued using the VSL.

The way that EPA uses academic estimates for the VSL in its calculations can be characterized as a type of benefit transfer. Of the 26 academic estimates for the VSL that feed into the composite estimate that EPA recommends using for policy analyses, 21 come from labor market studies that estimate a version of the wage-risk tradeoff in (6) (EPA 2010). In contrast, most of the people whose lives are estimated to have been saved from past reductions in air pollution do not face a wage-risk tradeoff because they are retired.

Focusing on adults aged 35 and over, Fig. 2 reports the fraction of deaths in the United States in 2010 by age category for fatal occupational injuries, particulate matter, and all causes combined. Statistics on all-cause mortality are from the US Centers for Disease Control and Prevention (Murphy et al. 2013); workplace fatalities are taken from the Census of Fatal Occupational Injuries; and estimated deaths induced by particulate matter are based on EPA's (1999) first prospective analysis of the Clean Air Act Amendments.[13] Overall, there were approximately 2.4 million deaths in the United States in 2010 and one tenth of one percent of them occurred in the workplace.

The distribution of deaths by age that EPA (1999) attributes to particulate matter closely matches the overall distribution of deaths by age from all causes in Fig. 2. About 90% of deaths from particulate matter and from all causes combined occur among people who are older than 54, and nearly 80% of all such deaths occur among people over age 64. In contrast, over half of all deaths on the job occur among workers who are younger than 54. Deaths on the job are of course concentrated among the working-age population whereas older adults are more vulnerable to short-term pollution spikes (Schlenker and Walker 2016). Hence, EPA's approach to valuating mortality reductions takes a composite VSL estimate based on wage-risk tradeoffs made by one subpopulation (working-age adults) and transfers it to an older subpopulation that is more vulnerable to air pollution and other sources of mortality risk. The assumptions underlying this transfer raise several issues for researchers to consider.

First, individuals' willingness to pay to reduce the risk of death from air pollution may differ from their willingness to pay to reduce the risk of death from on-the-job fatalities, about 40% of which are from car crashes and other transportation incidents. Second, even if

[13] Specifically, I take the mortality distribution by age from Table 5-4 of EPA's report.

people are indifferent to the nature of the risk so that Eq. (9) holds, the composite VSL for the aggregate working-age population may differ from the composite VSL for the population of retirees. Differences between VSL measures for the two groups may occur for multiple reasons. All else constant, theory and data suggest that the VSL should increase in wealth and in the length of one's remaining life span (Costa and Kahn 2004; Hall and Jones 2007). This is relevant because the average working age adult who died on the job in 2010 had approximately three times as many expected life years remaining (26.2) as the average senior whose death is triggered by air pollution (9.6) based on the measures of life expectancy used by EPA (1999). On the other hand, seniors are the wealthiest demographic group by age. Since these two mechanisms work in opposite directions, their net effect is unclear.

Given the importance of the VSL for federal policy, it would be useful to improve our understanding of how the VSL varies across demographic groups and risk contexts. Prior studies have independently developed VSL estimates using relationships similar to the tradeoffs highlighted by the first order conditions in (6), (7) and (8). Yet, differences in their empirical methods and study populations preclude comparing the results across demographic groups or testing the equality in (9).

Labor market studies estimating (6) tend to focus on working age adults and frequently exclude women (Mrozek and Taylor 2002), begging the question of whether male and female workers differ systematically in their willingness to substitute wage and occupational risk. Fewer studies have used neighborhood risks to develop VSL estimates from (7). Davis (2004) is a notable example. He leverages an unexpected cancer cluster to estimate the willingness to pay to reduce the risk of a statistical case of pediatric leukemia from data on housing price changes in two Nevada counties. The econometric identification strategy is compelling but the nature of the risk, the affected population, and the geographic scope of the study make it hard to directly compare the results with national VSL estimates from markets for labor and health care.[14] In the context of health care, Hall and Jones (2007) develop national age-specific evidence on the VSL for all-cause mortality from estimates for the marginal effect of medical expenditures on survival rates. They find that the marginal cost of saving an adult life declines with age. For example, they estimate that the marginal cost of saving a life among adults aged 30–34 was \$4.9 million in the year 2000 compared to \$790 thousand for adults aged 70–74. A large wedge between VSL measures for younger and older adults could explain why EPA's benefit transfer estimate in Fig. 1 is so much larger than benefit measures based on housing market choices. On the other hand, Hall and Jones's methodology is based on a social planner model that differs from the revealed preference logic suggested by the income-risk tradeoff depicted for an individual in (8). Because Hall and Jones focus on gross medical expenditures, as opposed to the individual's out-of-pocket costs, interpreting their marginal cost estimates as VSL measures requires assuming that a social planner (i.e. the federal government) *knows* the age-specific VSL and uses that knowledge to optimally design Medicare, Medicaid and all other aspects of our national health care system that ultimately determine the level of aggregate gross medical expenditures by age. While each approach is interesting and generates insights, much could be learned from developing research designs to systematically estimate VSL from national evidence on individual tradeoffs in different contexts.

[14] Additional assumptions are also required to interpret the change in housing prices as a measure of willingness to pay, as discussed in Sect. 5.

5 Are Capitalization Effects Sufficient Statistics for Benefit Measures?

Since people effectively choose their desired amenity levels when they choose a residential location, economists have frequently sought to use properties of spatial equilibria in housing markets to assess the benefits of improving environmental quality. Intuitively, if people care about a particular amenity, such as water quality, then they will be willing to pay more to live in neighborhoods with cleaner rivers and lakes. Following this intuition, if the quality of a certain neighborhood's rivers and lakes improves unexpectedly then, all else constant, we would expect people to bid up rents in that neighborhood, assuming housing supply is less than perfectly elastic.[15] Following language used by Chay and Greenstone (2005) analysts frequently refer to the change in rents that is *caused* by a change in an amenity as a "capitalization effect".[16] Over the past decade, numerous studies have leveraged exogenous sources of variation in environmental amenities to identify capitalization effects. Moreover, discontinuities in the structure of environmental regulations have enabled researchers to identify capitalization effects for policy-induced changes in environmental quality which have sometimes been interpreted as sufficient statistics for benefit measures and compared to costs to draw inferences on the ex post efficiency of regulations. Examples include retrospective studies of the Clean Air Act (Chay and Greenstone 2005), the Superfund program for cleaning up hazardous waste sites (Greenstone and Gallagher 2008) and the Clean Water Act (Keiser and Shapiro 2017). These studies collectively raise an important question. *Do econometrically credible estimates for capitalization effects of environmental policies provide credible measures of consumer welfare gains from those policies?*

Theory and empirical evidence suggest the answer is generally "No". Of course, one can write down a model in which capitalization effects reveal consumer welfare. Lind (1973) and Starrett (1981) demonstrate that under special assumptions about preferences and about the spatial dispersion of the amenity the size of the change in housing prices needed to achieve spatial equilibrium following an exogenous shock to an amenity will reveal households' precise WTP for that change. Turner's paper in this issue provides a similar demonstration. Relative to the conceptual framework outlined above, Turner adds a parametric representation of preferences and assumes that: (i) wages and job opportunities are unaffected by environmental regulation; (ii) there are no physical and psychological costs of moving; (iii) households are fully informed about the levels of all amenities; (iv) the equilibrium hedonic price function for housing is linear; (v) the shape of the price function is stationary; and (vi) housing in the study area is a weak complement for environmental quality in the sense that everyone living outside the study area is assumed to be indifferent to quality changes. These assumptions are common in housing market studies. They simplify data collection and econometrics, while creating a frictionless setting in which there is a simple mapping between the shape of a hedonic price function and MWTP among the population of consumers who are affected by the policy. However, evidence from the empirical literature suggests that each of these assumptions tends to be violated and, moreover, that the violations have first order impacts on estimates for consumer welfare.

15 This assumes weak complementarity holds in the sense that one must live near the improved rivers and lakes in order to derive utility from the quality improvement.

16 Hence, capitalization is used to describe comparative statistics for the transition from an initial equilibrium to a new equilibrium. I follow this convention despite the fact that it has potential to create confusion. Banzhaf (2015) explains that this convention differs from prior literature that used "capitalization" to describe cross-sectional correlation between prices and amenities in a single equilibrium.

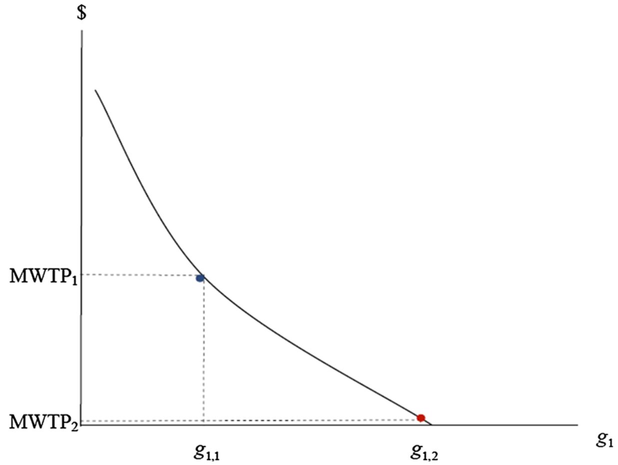

Fig. 3 A stylized example of the relationship between capitalization and welfare

5.1 Capitalization Effects May Not Reveal Average MWTP in the Study Area

To see the implications of the assumptions commonly invoked to interpret capitalization effects as benefit measures, consider some examples. First, in his investigation into the labor market effects of environmental regulations, Walker (2013) finds that the Clean Air Act Amendments led to more than \$7 billion in foregone earnings for workers at newly regulated plants. Second, Bayer et al. (2009) report that naively assuming migration is costless reduces their estimate of the marginal willingness to pay for air quality by more than 60%. This helps to explain the discrepancy in Fig. 1 between estimates for air quality improvements based on their results and those based on capitalization effects estimated by Chay and Greenstone (2005). Third, Pope (2008) and Armona et al. (2016) find that a significant fraction of consumers are misinformed about local amenities and housing price dynamics. For example, Pope (2008) demonstrates that a new policy mandating disclosure of publicly available information on whether a house is within an airport noise zone increased the implicit price of airport noise estimated from a hedonic price function by 37%. The effect would be zero if all households were fully informed. On the theory side, Ekeland et al. (2004) prove that equilibrium hedonic price functions are generically nonlinear and nonseparable. Kuminoff et al. (2010) demonstrate that the empirical bias from ignoring Ekeland, Heckman and Nesheim's result is likely to be similar in magnitude to the bias from ignoring the role of omitted variables in cross section regressions. Finally, in a boundary discontinuity study of the willingness to pay for public school quality in five metropolitan areas, Kuminoff and Pope (2014) demonstrate that ignoring temporal changes in the shapes of hedonic price functions results in understating households' MWTP for improving test scores by approximately 75%.[17]

Yet, capitalization effects are not necessarily uninformative. In certain cases they may identify an upper bound or a lower bound on consumer welfare. Suppose for the moment that migration and information are costless. Now imagine that an environmental policy substantially improves the quality of amenity g_1 from $g_{1,1}$ to $g_{2,2}$ without affecting wages. If

[17] Hedonic price functions may change shape over time due to changes in market primitives such as preferences, technology and institutions, macroeconomic shocks to wealth, or increased housing supply.

environmental quality is a normal good for people living in the affected area, then we would expect a representative consumer's MWTP to decline from MWTP_1 to MWTP_2, as shown in Fig. 3. A non-marginal change in quality may also change the shape of the equilibrium hedonic price function.[18] Nevertheless, Kuminoff and Pope (2014) demonstrate that the "gold standard" capitalization regression that uses an instrumental variable for the change in environmental quality identifies MWTP_2. This is important because capitalization studies typically approximate benefits by multiplying their estimates for MWTP by $\Delta g = g_2 - g_1$. Banzhaf (2015) proves that this approach yields a lower bound on a theoretically consistent measure of Hicksian equivalent surplus for the quality change in the special case where the capitalization regression identifies MWTP_2, wages are constant, and all households are freely mobile and fully informed about amenity levels before and after the policy.

Figure 3 illustrates why these issues matter for the way that capitalization-based benefit estimates are used to inform environmental policy. If $MWTP_2 \times \Delta g$ exceeds the cost of a policy, then the capitalization effect may be a sufficient statistic for policy analysis. It is sufficient in the sense that knowing the level of the true (larger) benefit measure would not change the conclusion that benefits exceed costs. Figure 1 suggests that Chay and Greenstone's (2005) estimate for the benefits of air quality improvements under the Clean Air Act Amendments may satisfy this criterion. On the other hand, if the cost of a policy exceeds $MWTP_2 \times \Delta g$ then the capitalization effect is not a sufficient statistic for policy analysis. In this case, the analyst cannot distinguish the hypothesis that $benefit > cost > MWTP_2 \times \Delta g$ from the competing hypothesis that $cost > benefit > MWTP_2 \times \Delta g$. This lower bound logic can also explain why estimates for the benefits of environmental improvements using different methods such as VSL, recreation demand or stated preference techniques may be large relative to estimates based on capitalization effects. Taking this idea to its logical extreme, if the WTP for past improvements in environmental quality are not fully capitalized into housing prices then total housing expenditures may, in principle, understate the welfare gains from past environmental regulations. This offers a second potential explanation for why EPA's estimate for the benefits of the Clean Air Act Amendments shown in Fig. 1 is similar in magnitude to total annual personal consumption expenditures on housing.

Moving from the regulation of air pollution to the regulation of hazardous waste and water quality, the distinction between capitalization and welfare is critical for interpreting results from Greenstone and Gallagher (2008) analysis of capitalization effects of the Superfund program and Keiser and Shapiro (2017) analysis of capitalization effects of the Clean Water Act. Both studies conclude that the costs of improving environmental quality exceeded the capitalized effects of those improvements in the prices of houses located near the improved sites.[19] The authors' conclusions make sense conditional on the assumptions embedded in their respective research designs, but one must be careful not to overextend their results to draw inference on the net benefits of the policies they study. It would be incorrect to interpret their findings as evidence that the costs of the Superfund program and the Clean Water Act exceeded the benefits they created for people living near the improved sites. Actual benefits could be much larger than Greenstone and Gallagher's and Keiser and Shapiro's lower bound

[18] Even if the quality change is small, concomitant changes in technology, preferences, and information may cause the shape of the price function to change. Indeed, such changes are likely to occur over the 10 to 30-year study periods that are common in empirical capitalization studies.

[19] Greenstone and Gallagher (2008) focus on median housing prices in neighborhoods around Superfund sites. Gamper-Rabindran and Timmins (2013) demonstrate that focusing on the median priced house may understate the capitalization effects of Superfund cleanups because those effects tend to be concentrated among houses at lower quantiles of the within-neighborhood distributions of housing prices.

estimates. Further, the benefits of those programs could extend beyond the authors' study areas.

5.2 Benefits May Extend Beyond the Study Area

When housing markets are used to estimate benefits of environmental policies the resulting benefit measures are only identified for people living within the spatial extent of the market defined by the researcher. Interpreting such measures as total benefits requires assuming that benefits do not extend beyond the study area. In the case of water quality improvements, for example, one must be willing to assume that people living far from the improved areas do not derive any benefits from taking recreation trips to water bodies in those areas. Likewise, one must be willing to assume a zero non-use value for people living outside the study area. As an extreme example of where these assumptions would be violated, consider the National Oceanic and Atmospheric Administration's natural resource damage assessment of the Deepwater Horizon oil spill. In a national contingent valuation study conducted as part of that assessment, Bishop et al. (2017) estimate that Americans would be willing to pay \$17.2 billion to avoid the environmental damages resulting from the spill. Likewise, of the \$520 million in estimated damages from lost recreation trips to shoreline areas on the Gulf of Mexico, approximately one third was attributed to people living outside states adjacent to the Gulf (English and McConnell 2015; Von Haefen 2016). A capitalization-based estimate for the effects of the spill in housing markets near the shoreline would have excluded these substantial damages. In cases such as these, where housing market capitalization effects may be inadequate for judging whether a policy has net benefits the challenge for researchers is to develop an empirical framework to directly measure consumer welfare.

6 Beyond Benefit Transfer

Looking ahead, it seems likely that the need to use benefit transfers to assess the benefits of environmental policies will decline over time. Researchers are increasingly able to access nationally representative administrative data sets that allow them to track individuals' spatial migration decisions, housing purchases, labor market participation, exposure to pollutants and health outcomes.[20] Together with increases in computing power and innovations in modeling, the current trend toward "big data" will hopefully improve our ability to estimate and validate models capable of estimating the benefits of national policies, moving closer to the type of benefit measure suggested by (10). Equilibrium models of household sorting behavior represent a step in this direction.

Roback (1982) proposed using nation-wide data on spatial equilibria in housing and labor markets to assess the benefits of policies that produce spatial differentiated changes in the quality of life. Since then, researchers have extended her framework to directly model how households sort themselves between and within metropolitan areas across the United States, taking account of heterogeneity in moving costs and job skills (e.g. Bayer et al. 2009; Cropper and Sinha 2013; Hamilton and Phaneuf 2015; Mangum 2015). These national

[20] In some cases, the use of administrative data sets may also pose new challenges in terms of sample selection. For example, Walker's (2013) use of LEHD administrative records to analyze the effects of air quality regulations on the labor market was necessarily limited to just four states and Ketcham et al. (2016) analysis of Medicare administrative records to assess the welfare effects of choice architecture polices proposed for health insurance markets necessarily excluded individuals who received low income subsidies because they faced a different choice structure that would have invalidated the research design.

models can potentially be used to predict the distributional benefits of prospective changes to environmental policy, embedding several of the features of the benefit measure in (10). However, there are also several potential areas for further research.

First, it would be useful to estimate the models using nationally representative samples. Prior studies have been "quasi-national" in the sense that they limited their analyses to major metropolitan areas and specific age groups. For example, Bayer et al. (2009) focus on household heads under the age of 35 living in 242 metropolitan statistical areas and Cropper and Sinha focus on recent movers between the ages of 26 and 55. While both studies have good reasons for limiting their samples, their calibrated models cannot be used to analyze welfare effects for demographic groups that may be of interest to policymakers such as seniors and people living in rural areas. Extending the estimation samples to be nationally representative would improve their relevance for evaluating federal policies.

Second, national sorting models have yet to be validated. They necessarily embed parametric assumptions on utility functions that impose restrictions on the tradeoffs that households would be willing to make between environmental quality and private goods, along with statistical assumptions made for econometric convenience. This can make it hard to tell what ultimately drives the results—assumptions or data. One way to resolve this uncertainty is to test a model's out-of-sample predictions. A well calibrated model of household decision making will make accurate predictions for how households respond to changes in incentives and constraints. Galiani et al. (2015) demonstrate this approach, showing that a model of household sorting within the Boston metropolitan area delivers remarkably accurate predictions for how households did or did not choose to adjust their residential locations in response to a novel policy. If national sorting models could be refined and shown to perform as well in similar validation tests, it would build confidence in using them to evaluate national environmental policies.

Third, it would be useful to extend current models to make prices endogenous. Chay and Greenstone (2005) and Walker (2013) demonstrate that national environmental regulations are sufficiently "big" to cause non-marginal changes in housing prices and wages, and these changes will feed back into household welfare. Sieg et al. (2004) develop a model of housing market sorting that is "general equilibrium" in the sense that it is capable of modeling price adjustments and Smith et al. (2004) implement the approach to analyze the spatial distribution of benefits from the Clean Air Act across the Los Angeles Metropolitan Area. The next challenge is to implement their logic at the national level.

Fourth, it would be useful for models of household decision making to address heterogeneity in their information and beliefs. There is abundant evidence that some consumers are not fully informed about local amenities (e.g. Pope 2008). However, there has been relatively little work on systematically incorporating this heterogeneity into models of decision making with respect to environmental amenities. It remains standard to assume that households' perceptions of environmental quality match the empirical information collected by analysts. Such assumptions can be tested by using survey instruments to elicit consumers' beliefs, which can then be incorporated into models of their choices to refine estimates for their preferences (Armona et al. 2016). Another approach is to use revealed preference tests and/or surveys to identify a subset of consumers who appear more likely to be fully informed and whose choices are therefore more likely to be informative about their preferences (Ketcham et al. (2016). Leggett (2002) illustrates how to use such information to refine estimates for the distribution of benefits from an environmental policy in settings where consumers have heterogeneous beliefs about environmental quality.

Finally, it may be useful to extend sorting models to incorporate the health care sector, directly parameterizing the conceptual framework outlined in Sect. 3. Health care represents a large and growing segment of the US economy (Hall and Jones 2007). Moreover, Finkelstein et al. (2016) suggest that there is significant spatial sorting on the basis of health as well as significant spatial variation in the cost of health care. This variation may be particularly relevant for understanding the effects of environmental policies, given the rapidly growing literature on the effects of pollution exposures on health and human capital (Graff-Zivin and Neidell 2013).

One specific reason for incorporating the health care sector is to improve our current understanding of how environmental policies affect health care costs. For example, Banzhaf and Walsh (2008) show that households collectively respond to increases in air pollution in their neighborhoods by moving out at a higher rate and that the increasingly polluted neighborhoods become poorer over time. Aside from income, it is less clear which demographic groups are moving, where they move to and how their migration patterns affect their health, medical expenditures and welfare. If higher pollution exposure worsens health and increases medical spending, then standard sorting models may understate the welfare losses experienced by stayers. Likewise, if movers experience fewer negative health shocks after moving to cleaner areas, standard sorting models may overstate their welfare losses from moving. To the extent that the health care costs of pollution exposures are concentrated among seniors and lower-income groups, it may also be important to understand how environmental quality affects taxpayer expenditures to support Medicare and Medicaid programs. A national sorting model capable of representing how an environmental regulation affects the economy through wages, housing prices and medical expenditures would represent a significant step toward developing a micro-founded, spatially explicit version of Hazilla and Kopp (1990) general equilibrium framework for environmental policy evaluation.

Another reason for incorporating health care as a margin of adjustment is that it may be helpful in identifying the VSL for seniors. Decisions about spending on medical care may be the most salient private market decision through which they can adjust their mortality risk. In principle, this could be done by replacing Hall and Jones (2007) social planner framework with a micro-founded model of individuals who are vulnerable to pollution-related health shocks and decide how much to spend on medical care based on their out-of-pocket costs, taking Medicare and Medicaid benefits as given from the consumer's perspective.

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