10-2-2009

Ecological Economic Applications for Urban and Regional Sustainability

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ECOLOGICAL ECONOMIC APPLICATIONS FOR
URBAN AND REGIONAL SUSTAINABILITY

A Dissertation Presented

by

Kenneth Joseph Bagstad

to

The Faculty of the Graduate College

of

The University of Vermont

In Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy
Specializing in Natural Resources

May, 2009

Accepted by the Faculty of the Graduate College, The University of Vermont, in partial fulfillment of the requirements for the degree of Doctor of Philosophy, specializing in Natural Resources.

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ABSTRACT

Urban and regional development decisions have long-term, often irreversible impacts on the natural and built environment. These changes impact society's well-being, yet rarely occur in the context of well understood economic costs and benefits. The cumulative effects of these individually small land use decisions are also very large. Ecological economics provides several frameworks that could inform more sustainable development patterns and practices, including ecosystem service valuation (ESV) and the Genuine Progress Indicator (GPI). This dissertation consists of a series of articles addressing urban and regional development from an ecological economic perspective, using GPI, ESV, and evaluation of tax and subsidy programs.

The GPI has been well developed at the national level but is of growing interest to stakeholders and citizens interested in better measuring social welfare at local and regional scales. By integrating measures of built, human, social, and natural capital, GPI provides a more comprehensive assessment of social welfare than consumption-based macroeconomic indicators. GPI's monetary basis allows these diverse metrics to be integrated, and can also facilitate intra- and inter-regional comparisons of social welfare.

Ecosystem services are also increasingly recognized as important contributors to human well-being, particularly in areas where they are becoming scarce due to rapid land conversion. Despite recent advances in measuring and valuing ecosystem services, they are often not considered in decision making because of both scientific uncertainty and the difficulty in weighing these values in tradeoffs. Techniques to speed the valuation process while maintaining accuracy are thus in high demand. As public recognition of the value of ecosystem services grows, ESV can serve as the basis for a variety of policy tools, from inclusion in traditional permitting or conservation easement programs to new programs such as payments for ecosystem services.

Ideally planners, citizens, and decision makers would better weigh the diverse costs and benefits of land use decisions as part of development and conservation planning. By quantifying changes in: 1) contributors to social welfare and 2) the value of ecosystem services across the urban-rural gradient, the GPI and ESV frameworks developed as part of this dissertation can thus be used to better inform local and regional policy and planning.

ACKNOWLEDGEMENTS

I would like to thank my advisor, Dr. Austin Troy, as well as my committee members, Drs. Marta Ceroni, Bob Costanza, and Jon Erickson, for their guidance throughout this work. I also thank my friends and family for their support throughout my graduate career, and especially my parents for instilling a love of learning that brought me through the finish line of this dissertation. Finally, thanks to the extended communities within the Gund Institute and Rubenstein School for providing the intellectual, moral, and social support that made UVM a great academic home for the last 3 ½ years.

TABLE OF CONTENTS

Page

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1: INTRODUCTION AND LITERATURE REVIEW

1. Urban and regional development in context

Urban and regional development decisions take place daily, yet rarely occur in the context of well understood economic costs and benefits. The cumulative effects of these seemingly small land use decisions can be extremely large. Between 1960 and 1997, urban land area in the United States grew by 157%. From 1982 to 2002, developed land increased by 47%. Accounting for population growth, per capita urban land in the U.S. rose by 73% and developed land by 18% during these time periods (Lubowski et al. 2006). Nowak and Walton (2005) predict widespread urban expansion by 2050, with the percent of the U.S. covered by urban land rising from 2.5% in 1990 to 3.1% in 2000 to 8.1% by 2050. Given such rapid changes to our natural and built environment, significant changes in social well-being are likely to accompany such urban expansion.

Planners and economists list numerous external costs of the low-density development that has typified recent decades. These costs include fiscally burdensome infrastructure (Esseks et al. 1999, Muro and Puentes 2004), automotive dependence, rising commute times, asthma and obesity, increased air and water pollution, and loss of farmland and open space (Burchell et al. 2005). Such quality of life implications associated with urban growth have often been described qualitatively. However, minimal effort has been made to quantitatively compare the tradeoffs inherent in these diverse metrics at regional scales. Furthermore, the costs of lost ecosystem services associated with the depletion of open space have only recently been appreciated and rarely quantified (Farber 2005). Beyond the urban fringe, more complete accounting for economic externalities can be valuable in comparing alternative agricultural management practices (Dale and Polasky 2007, Tilman et al. 2007), measuring the impacts of extractive industries (Anielski and Wilson 2007, Barbier et al. 2008), and documenting the effects of climate change (Schröter et al. 2005) or the value of ecosystems in mitigating the damaging effects of climate change (Costanza et al. 2008).

In the face of such imperfect information and externalities, it is reasonable to expect that many land use decisions will be economically inefficient (Daly and Farley 2003). Government-sponsored tax, subsidy, and insurance programs can also influence development patterns. These programs can be used to internalize externalities in cases where private and social costs diverge. Perverse subsidies, however, can increase divergence between private and social costs (Myers and Kent 1998). By identifying and measuring these externalities, ecological economists can provide guidance to help decision makers promote both sustainability and market efficiency.

In paper 1, “Taxes, subsidies, and insurance as drivers of United States coastal development,” we examined the effects of such policies on development patterns, particularly for coastal regions. We found that existing policies at the federal level are fragmented across different agencies, and often lack policy coherence with state and local policies. Various subsidies favoring development and extractive resource use in ecologically fragile coastal areas are damaging to valuable natural capital, while putting increasingly large populations and infrastructure at greater exposure to risk from coastal disasters. Through a more coherent policy approach, sustainable development patterns could be promoted for coastal zones to address issues of ecological sustainability, resource distribution, and economic efficiency.

Ecological economics provides several other tools that could potentially inform more sustainable development patterns and practices, including ecosystem service valuation (ESV, Costanza et al. 1997a, Daily 1997, National Research Council 2005) and the Genuine Progress Indicator (GPI, Daly and Cobb 1989, Anielski and Rowe 1999). This dissertation includes a series of articles addressing urban and regional development tradeoffs from an ecological economic perspective, evaluating development-related tax and subsidy programs and developing ESV and GPI tools for use at local and regional scales.

2. The Genuine Progress Indicator

The GPI and closely related Index of Sustainable Economic Welfare (ISEW) were originally developed as more comprehensive measures of economic well-being to compliment or replace Gross Domestic Product (GDP). GDP, a measure of overall macroeconomic activity, sums economic production activities within a national economy. GDP is also calculated for subnational political units, including states and metropolitan areas within the United States. Although the original developers of GDP accounting never intended it as a measure of social well-being, GDP is still used as such by many politicians, economists, business leaders, and the media. Anielski (2007) lists seven problems with using GDP as an indicator of well-being: 1) GDP counts all expenditures as contributors to well-being; 2) GDP excludes economically beneficial unpaid work; 3) GDP ignores the contribution of natural resources to well-being, and the costs of their depletion or destruction; 4) GDP ignores income distribution, poverty, and the costs of inequality; 5) GDP adds “defensive expenditures” that do not contribute to well-being; 6) GDP ignores the positive externalities resulting from investments in human and natural capital; 7) GDP fails to measure or account for social capital.

The ISEW and GPI are part of a family of indicators purported to measure human well-being in a more comprehensive manner than GDP (Böhringer and Jochem 2007). The ISEW/GPI begins with a measure of personal consumption, weighted to account for income inequality, and deducts or adds value for various monetized measures of built, human, social, and natural capital. This can be expressed in the form of the equation (adapted from Hanley et al. 1999):

$GPI = C_{adj} + G + W – D – S – E – N$ (1)

Where: C_{adj} = personal consumption adjusted to account for income distribution, G = growth in capital and net change in international position, W = non-monetary contributions to welfare (e.g., household labor, volunteer work), D = defensive private expenditures, S = depletion of social capital (e.g., cost of crime, family breakdown, lost leisure time), E = costs of environmental degradation, and N = depletion of natural capital. The inclusion of these components makes GPI better suited than GDP to addressing questions of distribution, societal well-being, and sustainability within the economy.

There has recently been a growing interest in developing meaningful quality of life measures at the community level, with the implicit goal of improving quality of life (Haggerty et al. 2001). Numerous communities have developed suites of quality of life indicators (Sustainable Measures 2006) yet face the problem of creating an index from indicators with dissimilar units (e.g., rates of farmland loss, voter registration, and crime). GPI overcomes the problem of non-comparability by estimating monetary values for various measures of built, human, social, and natural capital. Although a great number of national scale GPI studies have been undertaken, interest in developing local GPI estimates has grown recently. Internationally, GPI or ISEW studies have been conducted at local or regional scales in Australia (Lawn and Clarke 2006), Canada (GPI Atlantic 2005), China (Wen et al. 2007), Italy (Pulselli et al. 2006, Pulselli et al. 2008), and the U.K (Moffatt and Wilson 1994, Matthews et al. 2003, Jackson et al. 2006). In the U.S., GPI has been estimated sparingly at local scales, with Venetoulis and Cobb (2004) providing estimates for the San Francisco Bay area and Costanza et al. (2004) estimating GPI for Burlington, Chittenden County, and the state of Vermont. Going forward, consistent methods and theoretical framework for local GPI studies would be beneficial if this tool is to be more widely applied at local and regional scales.

The GPI was developed as a national-scale indicator, and local GPI studies face important limitations. These include limited data availability and the need for consistent data sources and methods, the fact that GPI does not account for cross-boundary impacts of manufacturing, energy production, or resource extraction, and the fact that local governments do not have full power to set policy related to the GPI's component indicators (Clarke and Lawn 2008). The first limitation can be overcome with careful and consistent data collection and management. The second limitation should be recognized, but can be addressed regionally by examining trends in GPI across urban to rural environments. As for Clarke and Lawn's third limitation, state and local governments in the U.S. do have important policymaking powers in regards to land use planning, energy use, and other relevant GPI components. Finally, given GPI's value as a "debunking index" that exposes the limitations of GDP (Ziegler 2007), its use at local scales is just as relevant as at national scales. Given the lack of dialogue in the U.S. about alternatives to GDP since the mid-1990s (Cobb et al. 1995), this discussion may be more fruitful at the local level than the national level.

In paper 2, "Opportunities and challenges in applying the Genuine Progress Indicator/Index of Sustainable Economic Welfare at local scales," we further developed methods to measure the GPI at local and regional scales. We described the benefits and difficulties of measuring GPI locally, and provided comparative GPI estimates for seven counties in northern Vermont. We found that although local data quality has been problematic in the past, it has improved to the point of enabling more reliable GPI analysis from 1990 onward. In Vermont, per capita GPI was greatest in the wealthiest county (Chittenden) in the study area, and lowest in less affluent Northeast Kingdom counties (Caledonia, Essex, Orleans). Like Costanza et al. (2004), we found per capita GPI to rise continually throughout the study period, unlike U.S. GPI, which has flattened out or declined in recent years.

In paper 3, “The Genuine Progress Indicator as a measure of local and regional economic welfare: A case study for Northeast Ohio,” we calculated and compared GPI values for a 17-county region in Northeast Ohio. We used the most rigorous indicator data yet compiled for a U.S. local GPI study, while enabling comparisons of economic welfare as measured by GPI across urban, suburban, and rural counties. We found per capita GPI to be greatest in the wealthiest suburban counties and lowest in the cities of Cleveland and Akron. Per capita GPI declined in 9 counties, the state of Ohio, and Akron and Cleveland from 1990-2005, and grew in 8 Northeast Ohio counties. The relative growth in personal consumption relative to other environmental and social costs dictated whether per capita GPI grew or declined in a given region over the study period.

3. Ecosystem service valuation

ESV is increasingly being used to estimate the flow of economic value provided by natural ecosystems to people at regional scales (e.g., Kreuter et al. 2001, Wilson and Troy 2003, Costanza et al. 2006). Given recent trends in urban expansion, ESV can better inform local and regional land use and conservation decisions, but only if appropriate tools can be developed to apply ESV with increasing speed, accuracy, and transparency. Ecological and socioeconomic systems are inherently complex and unpredictable, which is one reason that ecosystem services are difficult to map, assess, and value. Given these difficulties, the practice of value transfer (Brookshire and Neill 1992, Wilson and Hoehn 2006) has grown in popularity to speed the mapping and valuation process for regional ESV applications. Value transfer uses economic values from studies conducted at a past study site, then applies these values to a policy site. Value transfer can be of two types – point transfer, which directly applies values from study site to policy site, and function transfer (Loomis 1992), where a mathematical function is applied to account for differences in resource characteristics, geographic setting, and the constructed market. Regrettably a lack of quality meta-analyses, caused in turn by shortages of primary valuation studies, limits the opportunities to use function transfer for most ecosystem types. For many ecosystem services and land cover types, there are also shortages of primary studies to provide even point estimates for value transfer. Boyle and Bergstrom (1992), Desvousges et al. (1992), Brouwer (2000), and Spash and Vatn (2006) note that basic equivalence of the population, institutional setting, environmental resource, and constructed market characteristics is needed for sound value transfer.

A major weakness of most recent value transfer studies is their application of a single value coefficient for broad land cover categories (e.g., “forests”, “wetlands”) that include wide contextual variation, a violation of the equivalence requirement noted above. To improve the accuracy of regional ESV estimates, land use categories should be made more precise by taking their ecological and socioeconomic context into account. GIS data can often provide spatial information to bridge this gap.

Ecological setting is an important contributor to value. Urban form has important effects on ecosystem processes (Alberti 2005). Using common conventions (de Groot et al. 2002, Millennium Ecosystem Assessment 2005), ecosystem processes provide the supporting services that contribute to regulating, provisioning, and cultural ecosystem services. Ecosystem processes can be predicted with some accuracy based on urban form, by considering patch size, shape, and distribution, and disturbance processes. Urban form, land use intensity, heterogeneity, and connectivity are key measures that affect ecosystem functions like net primary productivity, biodiversity, soil quality, runoff, sedimentation rates, nutrient cycling, and natural disturbance processes (Alberti 2005), although many of these relationships are still poorly understood.

An example of the importance of socioeconomic setting is the "marginal value paradox" (Mitsch and Gosselink 2000), where the value of each unit of open space increases as human populations grow and open space shrinks. However, if human use of the open space continues to intensify, it can overwhelm the capacity of the ecosystem to provide services, leading value to decline sharply. The paradox is that open space becomes more critical to human well-being the scarcer it becomes on a local scale, but its value can ultimately be destroyed by overuse. Generally, this principle implies a higher per-unit value for urban ecosystems and lower value for rural ecosystems (Fausold and Lilieholm 1999). For example, it has been frequently noted that the marginal value of an acre of Central Park is worth far more than that of an acre of parkland in rural upstate New York. Although time consuming to measure, preferences for natural capital also likely differ among different socioeconomic groups and along the urban-rural gradient (Grove et al. 2006, Mulder et al. 2006).

While many studies have closely examined the ecological "supply side" of ecosystem services, fewer have focused on the societal “demand side.” This may be in part due to the abstract nature of the MA classification system for ecosystem services. Several authors have recently argued for revised conceptualizations of ecosystem services that place human beneficiaries and the benefits they receive as the primary perspective from which to conduct ESV (Boyd and Banzhaf 2007, Wallace 2007, Fisher and Turner 2008). Mapping beneficiaries and the provision of ecosystem services offers a first step toward estimating spatial flows of ecosystem services, an important advance that researchers have yet to achieve (Tallis et al. 2008).

Finally, neoclassical economic valuation theory, which operates based on marginal values, is also problematic for ESV for several reasons. Values are typically estimated using stated or revealed consumer preference. Consumer preferences are usually expressed in the face of ignorance and imperfect information of how ecosystem services are supplied and delivered. Gowdy and Mayumi (2001) describe the problems in consumer choice theory that forms the basis for neoclassical economic valuation, including that of ecosystem goods and services. The thresholds, non-linearities and irreversibilities that exist in land use decisions are also problematic from a marginal valuation perspective (Arrow et al. 2000, Farber et al. 2002, Limburg et al. 2002, National Research Council 2005, Farley 2008). Many consumptive land use decisions are essentially irreversible. For example, the choice to build a subdivision precludes the future use of such land for its past open space use, such as farmland, grassland, or wetland. Marginal valuation is poorly suited to the presence of such irreversibility. It also performs poorly in the face of ecological thresholds (Muradian 2001, Resilience

Alliance and Santa Fe Institute 2004, Walker and Meyers 2004). Perhaps the best understood such threshold in urban systems is the impact of watershed level impervious surface on watershed health. Beyond a certain impervious surface threshold, an "urban hydrograph" develops, leading to reduced groundwater recharge, increased flashiness of the hydrograph, with larger flood peaks and reduced dry-period baseflows, water pollution, stream downcutting and geomorphic instability, and degradation of the aquatic biota (Paul and Meyer 2001, Center for Watershed Protection 2003). The regulation of hydrologic flows is thus likely to be an important ecosystem service in densely populated settings (Pagiola and Ruthenberg 2002). Tools to improve the accuracy and quality of ESV in urban and regional settings should seek to address these questions about ecological and socioeconomic setting, preferences, thresholds and irreversibility in valuation, and service flows between ecosystems and beneficiaries. Further, static ESV across large regions faces the limitation that valuation estimates are typically derived from marginal values. Extending these values across a large landscape assumes that ecosystem service users would be willing to pay (or be compensated) a certain economic value for their loss. Yet the reduced supply of ecosystem services in the event of such a loss would lead to a change in the value of natural capital, rendering the original values incorrect (Troy and Wilson 2006). This is one critique originally directed at Costanza et al.'s (1997a) global ESV estimates, but it is relevant whenever local economic values are applied across large parts of the landscape.

In paper 4, "Context matters: Applying ecological and socioeconomic criteria for improved ecosystem services benefits transfer," we demonstrated the need to incorporate landscape scale ecological and socioeconomic context when valuing ecosystem services at the regional level using value transfer. Past regional valuation exercises have been based on subjective, *ad hoc* land use-land cover (LULC) typologies for ecosystem services. We use two methods to illustrate the importance of developing more systematic and precise land cover typologies for ecosystem services assessment. First, we catalog the ecological and socioeconomic contextual variables that have been used by past meta-analysts. These authors have identified numerous contextual variables that are both theoretically important and statistically significant in influencing ecosystem service values, but have not yet developed a systematic way to describe, catalog, and use these contextual variables. Second, we reexamine primary studies from a past value transfer exercise for the state of New Jersey, comparing value ranges for a high precision and a low precision LULC typology. We find that precise typologies that better account for ecological and socioeconomic context produce narrower value ranges, increasing the potential accuracy and value of benefit transfer results.

In paper 5, "From ecosystems to people: Characterizing and mapping the beneficiaries of ecosystem services," we built the case for using human beneficiaries as central "accounting units" when measuring and mapping ecosystem services. The "supply side" of ecosystem services – the ecosystems providing values to humanity – are relatively well researched and are often mapped using spatial ESV approaches. However, the "demand side," or human beneficiaries of ecosystem services are often less well understood. We identified benefits and beneficiaries for two ecosystem services – carbon sequestration and storage and aesthetic value, each characterized by different groups of beneficiaries and means of benefit flow from ecosystems to beneficiaries. We then demonstrated how the ARtificial Intelligence for Ecosystem Services (ARIES) system can map ecosystem services supply and demand, extending the spatial mapping of ecosystem service provision undertaken by past studies. The resulting beneficiary maps can be combined with provision maps and models to describe how benefits flow from ecosystems to beneficiaries. These provision, use, and flow maps can greatly advance both the science and policy applications for ecosystem services.

4. Conclusions and connections

Although they are distinct approaches for monitoring sustainability and social well-being, GPI and ESV share a theoretical link in the field of ecological economics. As opposed to the neoclassical economic goal of maximizing the sum of producer and consumer surplus, often measured in the aggregate by GDP, as the ultimate desirable end, ecological economics views the contributions of natural, social, human, and built capital as non-substitutable compliments needed to produce social well-being or quality of life as an ultimate desirable end (Ekins 1992, Costanza et al. 1997b). The GPI incorporates these "four capitals" as its basic components, including natural capital measures of forest, farmland, and wetland loss. As such GPI implicitly recognizes the connection between natural capital and societal well-being. As increasing population, affluence, and technology continue to make natural capital more scarce, ESV has become increasingly viewed as an important tool for informing environmental, economic, conservation, and land use policy. ESV also offers the opportunity to inform estimates of GPI and “Green GDP” (Boyd 2007).

The vast majority of land use decisions ignore many of the negative externalities of decentralized growth. Yet in recent years, citizens in regions experiencing rapid open space loss have overwhelmingly supported referenda to publicly acquire more open space (Kline 2006, Nelson et al. 2007). This indicates a public perception of the value of preserving increasingly scarce natural capital as an important contributor to quality of life. As public recognition of the value of ecosystem services grows, ESV can serve as a basis for a wide variety of policy tools, from inclusion in traditional zoning and permitting or conservation easement programs to new programs such as payments for ecosystem services (Levitt 2005, Salzman 2005). Ideally planners, citizens, and decision makers would better weigh the diverse costs and benefits of land use decisions as part of the process of development and conservation planning. By exploring changes in: 1) contributors to well-being and 2) the value of ecosystem services across the urban-rural gradient, GPI and ESV tools can be better developed to help inform local and regional development and conservation planning process.

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CHAPTER 2: TAXES, SUBSIDIES, AND INSURANCE AS DRIVERS OF UNITED STATES COASTAL DEVELOPMENT[1]

Abstract

Ever-increasing coastal populations in the United States and worldwide are putting growing quantities of people and property at risk due to coastal disasters. At the same time, poorly planned development policies and practices erode the natural capital of coastal regions, eliminating existing landscape protection from intense wind and waves. Government tax, subsidy, and insurance policies can encourage or discourage particular forms of development. In the U.S., there is no consistent set of incentives or disincentives for coastal development, and many programs have ambiguous or contradictory goals. Federal programs are highly fragmented, being administered by a variety of government agencies. State and local governments can also implement policies to improve coastal disaster protection, but often fail to do so. In other cases state and local policies designed for local economic growth work against the goals of federal policy, increasing flood damage risk while relying on federal aid once disaster does strike. These programs frequently lead to perverse subsidies, where economically inefficient policies degrade natural capital and foster economic inequality. In this study, we evaluate the existing tax, subsidy, and insurance structures that led to coastal development patterns on the U.S. Gulf Coast over the last sixty years,

[1] Bagstad, K.J., K. Stapleton, and J.R. D'Agostino. 2007. Taxes, subsidies, and insurance as drivers of United States coastal development. Ecological Economics 63: 285-298.

and propose alternative policies that could create a more sustainable, just, economically efficient, and storm-adaptive region.

Keywords

Taxes, subsidies, National Flood Insurance Program, coastal development, Hurricane Katrina

1. The general role of taxes, subsidies, and insurance in coastal development

Government intervention in the market, particularly through taxes, subsidies, and insurance, plays a major role in influencing development patterns worldwide, and especially in the United States. When used inappropriately, these measures can distort true costs and incentives for particular economic sectors while degrading economic, social, and environmental well-being; however, they can also be designed in a manner that enhances economic, social, and environmental quality.

Land and property tax incentives at the local level, particularly tax breaks or subsidized infrastructure, are often used to entice development in particular regions or economic sectors. At the national level, certain tax breaks can encourage or discourage development of new land versus redevelopment of underutilized land, or provide incentives or disincentives to restore, rehabilitate, or improve existing structures. Although seldom implemented, taxes on rents from land and natural resource extraction show promise in improving the efficiency of these resources' use (Daly and Farley 2004).

A subsidy is a "payment by a government to an individual or firm, the intent of which, theoretically, is to decrease the divergence between social costs and benefits – to internalize externalities" (Costanza 2001). Subsidies can produce a socially desirable outcome in numerous instances, such as for a public good that would be produced at a sub-optimal quantity if its provision were left solely to the market, or in the case of a socially desirable industry or technology that may require an initial public investment to become economically competitive with established industries. Subsidies can be "on-budget" – programs that transfer money between government, industry, and taxpayers, leading to a change in the government budget (e.g., a tax cut or increase), or "off-budget", when there is no direct monetary transfer but a change in assets or liabilities (e.g., policies that reduce or increase stocks of natural, human, social, or built capital) (van Beers and van den Bergh 2001).

Perverse subsidies, however, <u>increase</u> the divergence between private and social costs and benefits. Perverse subsidies are both economically inefficient and environmentally or socially damaging, and are extremely abundant in today's economy (Myers and Kent 1998). In their most severe forms, perverse subsidies can have numerous damaging effects on public welfare – increased poverty and economic inequality; economic instability; economic colonialism (one nation, state, or community extracting wealth from another without reinvesting in the local economy); reduced government vitality and responsiveness; and erosion of democracy. Templet (2001) discusses this cycle for Louisiana, where perverse subsidies to industry increased pollution, corporate profits, and income disparity, leading to a downward spiral of rising poverty and concentration of political power (Figure 1).

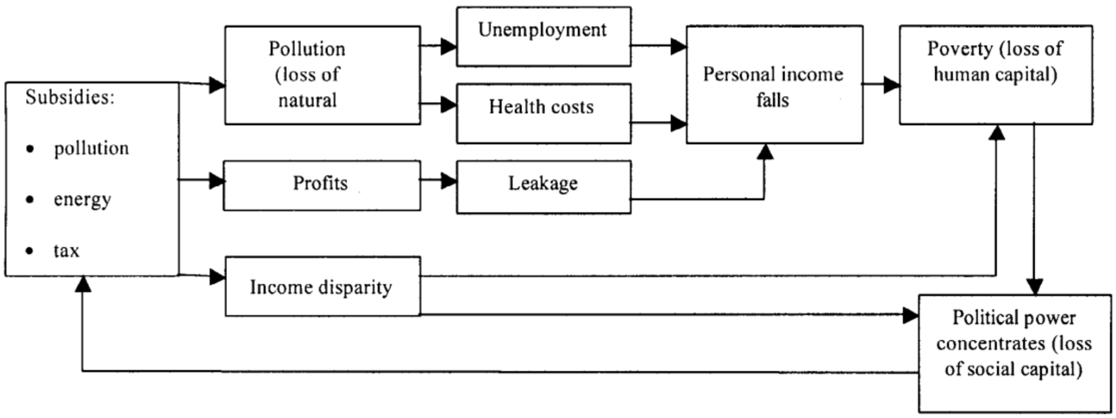

Figure 1: Effects of perverse subsidies on the economy (Templet, 2001).

Subsidized insurance allows landholders to develop areas that the market alone might otherwise deem too risky for construction – floodplains, coastal zones, and areas prone to earthquakes, mudslides, or wildfire. By matching pooled risk to premiums, private insurers maintain the viability of their industry. Government-subsidized insurance, through the National Flood Insurance Program, was originally intended to reduce flood zone development and risk. It has instead encouraged risky development while providing a subsidy to coastal and floodplain developers, repetitive loss property owners, and the private insurance industry. The decision to provide insurance and other reconstruction aid by federal, state, and local governments can lead to development of places that would otherwise be economically unsuitable for construction.

Tax, subsidy, and government-sponsored insurance programs distort market outcomes; in this way, they can lead to economic inefficiency. A key question with these policies, then, is whether they provide accompanying economic, social, and environmental benefits to justify their existence. On the U.S. Gulf Coast, particularly in Louisiana and Mississippi, the development patterns that arose prior to Hurricane Katrina took place under the influence of a variety of tax, subsidy, and insurance programs, many with ambiguous or conflicting goals. Several factors, including a sense of entitlement to subsidies, a "quiet" hurricane cycle from approximately the 1950s-1980s, and skyrocketing coastal populations created a climate that turned natural disaster into human tragedy in August and September of 2005. Here we evaluate the existing U.S. tax, subsidy, and insurance programs relevant to coastal development, and evaluate these policies and possible alternatives through the ecological economic criteria of sustainable and desirable scale, just distribution, and efficient allocation.

At the state level, we focus primarily on Louisiana, although in 2005 Hurricanes Katrina, Rita, and Wilma caused severe damage throughout the Gulf Coast. For comparison, we present estimates for each state of built, human, social, and natural capital for those counties directly adjacent to the coast, as well as those along Lake Pontchartrain in Louisiana (Table 1). We use population density as a proxy for built capital. Population, poverty, education, and income inequality act as human and social capital variables. For natural capital, we estimate ecosystem service product for the coasts using spatial data for 2001 from NOAA's Coastal Change Analysis Program, with areas multiplied by ecosystem service values from Cosatanza et al. (1997).

Generally, the coasts are more densely settled and poorer than the national average. Education levels are relatively close to the national average, but are notably lower along the Texas coast. Income inequality is slightly less for coastal counties than the state average but reflects state-level trends. In this respect, inequality is greatest in Louisiana than any of the Gulf states. Ecosystem service values are highest in Louisiana, due to its abundant but threatened coastal wetlands.

Table 1: Comparison of coastal counties for U.S. Gulf Coast states

	Texas	Louisiana	Mississippi	Alabama	Florida	U.S.*
Population	1,741,543	1,989,998	376,461	564,013	13,596,229	296,419,494
% of population living on coasts	8%	44%	13%	12%	76%	53%**
Population density (persons/km^2)	60	73	81	77	175	32
Per capita income	$14,319	$18,452	$17,897	$18,230	$22,329	$21,587
Population below poverty line	12.4%	16.9%	15.9%	16.1%	13.0%	12.5%
Over 25 with HS diploma	60%	76%	80%	78%	74%	80%
Over 25 with college degree	14%	21%	18%	20%	26%	20%
Gini coefficient	0.460	0.480	0.427	0.460	0.472	0.463
Coastal ecosystem service product/yr (billion 2000 dollars)	$17.1	$50.1	$3.8	$4.8	$8.8	$112.6****
Coastal ESP/km^2-yr (2000 dollars)	$583,775	$1,825,768	$812,933	$661,653	$684,475***	

* U.S. values are for the entire nation unless otherwise noted.
** This uses a more broadly defined coastal zone (Crossett et al. 2004).
*** NOAA's Florida land cover data includes only the six westernmost coastal counties in the state.
**** This value was calculated using 1 km^2 land cover data (Perez-Maqueo et al. 2007) and uses a more broadly defined coastal zone, so is not directly comparable to other state values.

2. Insurance, disaster relief, and mitigation

A broad spectrum of federal programs, along with state and local ordinances, provide tax and subsidy incentives that may encourage or discourage different development patterns. While a full evaluation of these programs is beyond the scope of this paper, we briefly describe relevant programs in Sections 2-3. We first consider the National Flood Insurance Program, a particularly important component of national floodplain and coastal policy, as well as Stafford Disaster Relief, and mitigation measures common to these two programs. A discussion of other tax and subsidy programs follows in Section 3. Unfortunately, while dollar values for these programs are available in many cases, many sources did not include the appropriate year for their dollar values, making comparison or summing of values problematic. Values also were not always available at consistent scales (nationally, by state, or coastal zone). Our approach was to provide dollar values wherever possible, with the appropriate year when it was provided by the source.

2.1 The National Flood Insurance Program

The National Flood Insurance Program (NFIP) was established in 1968 as an economic means to address the insufficient floodplain management practices of levee, dike, and dam construction that characterized the preceding decades. Its creation was also at least partly in response to damage to New Orleans by Hurricane Betsy in 1965. By the 1960s, there was growing recognition that despite decades of spending on levee construction and other structural flood control measures, the nation's flood damage risk had not been reduced. This led to support for the NFIP. The NFIP was expanded in 1973 to include coastal hazard zones (as part of the 1973 Flood Disaster Act) and was amended again in 1994 and 2004. The NFIP has the ability to pay about $700 million per year (which it defines as a historical average loss year). Since 1969, the NFIP has paid $11.9 billion that would have otherwise come from disaster relief payments (National Flood Insurance Program 2002).

Theoretically, for the NFIP to function properly, three important assumptions must be met (Krutilla 1966). First, the buyer and builder of floodplain or coastal property must know the costs of flood insurance. Second, enrollment in the insurance program be compulsory for properties located in flood zones. Third, the flood insurance premium must be tied to the risk of paying the claim, so the system is actuarially sound and aligned with social costs of floodplain development. An ecological economics perspective also requires a fourth assumption – that development occurs on a sustainable scale and does not negatively impact distribution and allocation.

There is substantial evidence, both anecdotal and quantitative, that the first two assumptions are rarely met. Chivers and Flores (2002) found that 70% of purchasers learned of the flood insurance rates at closing, and 21% learned after that time. Evidence also suggests that undeveloped flood-prone land sells at a discount (due to perceived flood risk) but developed flood-prone land sells at a premium, creating an incentive for developers to build in flood-prone areas in order to maximize profit (Holloway and Burby 1990). Although developed floodplain property does sometimes sell at a discount (reducing development incentives) and elevation requirements do reduce damage, ongoing floodplain development continues to increase overall risk due to floods and storms (Holloway and Burby 1993). Most studies agree that a combination of improved local land use planning, actuarially sound rates, mandatory participation in the flood insurance program, and improved information to prospective buyers would strengthen the NFIP. Weak building code enforcement has also plagued flood risk areas, particularly in Florida, exacerbating the challenges of reducing premiums paid by the NFIP (Kunreuther 1996). Regulations mandating insurance are also routinely avoided because local governments are responsible for enforcement and policies are often allowed to lapse because of a lack of oversight.

The third assumption, that the system is self supporting and actuarially sound, is also not met. George Bernstein, the first Flood Insurance Administrator, testified to Congress in 1973 that "the combination of effective land use controls and full actuarial based rates for new construction… makes the NFIP an insurance program rather than a reckless and unjustifiable giveaway program that could impose an enormous burden on the vast majority of taxpayers". FEMA asserts that the NFIP is designed to be self-sufficient – a claim that today is hard to justify upon examining the program's performance. Before 2005, various estimates put program losses at $450 million annually (Gaul and Wood 2000). Congress also forbids the program to charge enough to cover catastrophic losses (hence the unsound rates for the riskiest participants). In doing so, they leave the program vulnerable to massive losses, as the Katrina cleanup is proving. Estimated NFIP payments from damage induced by Hurricane Katrina are approximately $23 billion. Prior to Katrina, the NFIP had authority to borrow up to $1.5 billion from the Treasury Department, which must be repaid with interest. Katrina was the first time the NFIP's financial obligations surpassed this ceiling; the borrowing limit was subsequently raised to $20.7 billion with passage of the NFIP Enhanced

Borrowing Act of 2006. The $23 billion in estimated claims from the 2005 hurricane season is more than the total amount paid in claims by the NFIP through its entire history. In order to internally absorb catastrophic risk such as the 2005 hurricane season, revenues from policyholders would have to double. FEMA's own study of the economic effects of removing subsidies predicts that the average premiums for residential properties exposed to considerable flood risk would likely increase from $585 to about $2,000 (Price Waterhouse Coopers LLP, 1999).

Left to the market, flood insurance would not be offered or at best would be offered at far higher rates. For example, the Office of Technology Assessment (1993) estimates that premiums run approximately $800/yr in high-risk coastal areas, while private insurers would need a $12,000/yr premium to maintain a viable private program (Finegan 2000). Insurance companies cannot underwrite such predictable and catastrophic loss to large areas at rates that would make development feasible. The concept of pooling risk is not efficient when the only purchasers of a policy are those at great and predictable risk.

Unlike private insurance, the NFIP also pays claims multiple times for the same property, and does not raise rates with additional claims, which encourages rebuilding in the most flood prone areas. When disaster strikes, developers are able to buy up large amounts of land at steeply discounted rates, knowing they can rebuild and sell that property at rates that do not reflect the site's propensity for flooding. This rebuilding process costs the NFIP hundreds of millions of dollars each year. Repetitive loss properties account for about 2% of policyholders (approximately 82,000 of 4.1

million participating households) but almost 30% of all claims, totaling over $200 million per year. Absent from the NFIP's authority is the ability to condemn houses or require they be moved. Those decisions remain in the hands of local officials. If the NFIP can demonstrate that damage has reduced the market value of the property by at least 50%, it can require that owners elevate the structure when they rebuild. It cannot, however, require the building to be moved or reject reinsuring the property upon rebuilding.

The 2004 Flood Insurance Reform Act sought to address the problem of repetitive loss properties. A "three strikes and you're out of the government's pocket" program was established to deal with properties that incur three claims of $3,000 or more with cumulative claims damages of $15,000 or more, making them no longer eligible for NFIP insurance reimbursement for losses. Because so many participating NFIP homeowners were grandfathered into the program at subsidized rates, they have minimal incentive to conform to stronger recent floodplain building codes, leading to the repetitive loss cycle. Grand Isle, Louisiana's only inhabited barrier island, is a prime example of the problems of repetitive loss and perverse economic incentives provided by the NFIP. Grand Isle has been hit by 50 major storms in the past 130 years. According to Tulane University's Oliver Houck (Burdeau 2004), the total federal spending in Grand Isle amounted to $439,000 per home. Subtracting the many vacation homes increases the subsidy to $1.28 million for each of its 622 year round residents. Houck concluded that the government is funding high-risk coastal development, and suggested ending this subsidy, buying up flood-prone areas, and moving people back to low-risk zones.

Generally, private insurers have a strong interest in risk prevention and minimization, and devote considerable resources to disaster planning and mitigation. Well before Katrina, there was substantial concern among the insurance industry that exposures are increasing due to a relocation of large numbers of wealthy people to coastal areas. With short-term memory for disasters and short economic horizons, particularly for real estate speculators or transient residents who stay in their home for only several years, there is a strong disincentive to buy insurance or make structural improvements to mitigate for potential disaster loss (Kunreuther 1996). Given the statement by reinsurers that $45-50 billion in claims would lead to major insolvencies in the insurance industry (Kelly and Zeng 1999), perhaps it is fortuitous for these industries that most damage from Katrina came from flooding, and not wind damage.

Finally, perhaps the largest fault of the NFIP is that it encourages development in environmentally sensitive areas, decreasing the likelihood of development at a sustainable scale. The program externalizes the risk associated with building while imposing the added social cost of foregone ecosystem services. In providing flood protection, even the best structural measures usually fail as sufficient substitutes for intact natural capital.

The NFIP currently fails all four of the aforementioned requirements to properly function. Buyers and sellers have asymmetric information about the actual cost of flood insurance. Though nominally mandatory, many people avoid maintaining coverage, leading to moral hazard. The program is not actuarially sound, as a substantial number of policy holders do not pay premiums commensurate with risk. Finally, the program acts as a subsidy to encourage unsustainable development in high risk areas, depleting natural capital and externalizing the inherent risks of building in flood zones.

2.2 Stafford Disaster Relief

The Stafford Disaster Relief and Emergency Assistance Act was passed in 1988 as an amended version of the 1974 Disaster Relief Act. It was amended again in 2000 as the Disaster Mitigation Act. The Act's goal is to assist victims of natural disasters; it includes direct grants to victims (families and individuals) and assistance to communities to rebuild infrastructure. The Act is intended for use in areas where disaster recovery efforts require more resources than state and local governments can provide; following a Presidential declaration, disaster relief funds are distributed. Legally, state and local entities are required to share at least 25% of disaster relief costs, with the Federal Government paying 75%, but this contribution by state and local governments has declined in recent years. Starting with Hurricane Hugo in 1989, the federal share has often risen to 90-100% (Boswell et al. 1999). In Louisiana, the state's initial response when presented with their share of the Hurricane Katrina relief costs ($3.7 billion of an initial $41.4 billion in disaster relief, a 9% cost share by the state) was to immediately seek to reduce the state's share (Office of the Governor 2005). Most states would clearly struggle to pay such an unexpected bill; however, over-reliance on federal assistance reduces the incentive for state and local government to make strong commitments to disaster mitigation. This again encourages development of high-risk and environmentally sensitive areas.

Between 1990 and 2003, over $42 billion, an average of $3 billion per year, was spent on Stafford Disaster Relief (including coastal disasters and other Presidentially-declared disasters; FEMA 2004). In the 1980s, $3.9 billion was spent on declared disasters and emergencies; this figure grew to $25.4 billion in the 1990s (Platt et al. 2002). There is widespread concern that booming coastal population, changing climate, and sea level rise could dramatically increase costs of the program beyond today's levels. FEMA has compiled disaster relief data by state and year since 1988; for Louisiana, this totaled $273 million (in 1996 dollars; USFWS 2002).

Unfortunately, politicians appear to occasionally manipulate the disaster relief system. In a study spanning the years 1991 through 1999, Garrett and Sobel (2002) showed that states with political importance to the President (especially "swing" states in election years) have more disasters declared, and that states with greater representation on congressional FEMA oversight committees receive greater disaster relief funding for their state. Since the President has sole authority to declare disasters, and is given open-ended criteria for their declaration (particularly following the 1988 amendments), the average number of declarations per year increased from 25 for the period of 1983-1988 to 41 for 1989-1994. When they compared expected and observed relief spending, Garrett and Sobel determined that nearly half of all disaster aid is estimated to come from political "need".

2.3 Mitigation: relocation and alternatives

Through FEMA's Flood Mitigation Assistance (FMA) program (and its predecessors, the Upton-Jones and Section 1362 programs prior to the 1994 Flood Insurance Reform Act), the Federal Government can fund property acquisition and relocation in an effort to reduce settlement pressures in high risk areas. These programs have been used sparingly, particularly following FEMA's absorption into the Department of Homeland Security. Relocation was most commonly applied following the extensive 1993 and 1995 floods on the Mississippi River (Kunreuther et al. 1998, pg. 143). Dramatic cost savings from these programs were achieved in some cases – for example, in St. Charles County, Missouri, over 900 families were relocated to higher ground following the 1993 flood. Disaster relief costs from the federal and state government in 1993 were $14.2 million; moving the residents cost an additional $14.6 million. However, with these residents gone by 1995, that year's flood resulted in disaster relief costs of only $216,000 (Stonner 1999). In other cases, entire towns, such as Valmeyer, Illinois, decided to move from the floodplain to higher ground, breaking an ongoing cycle of flood damage and government relief spending. There is some concern that given the high cost of coastal property, these programs may work more effectively in floodplains (Office of Technology Assessment 1993). Still, when evaluated from a long-term cost perspective, a one-time relocation is clearly cheaper than an ongoing cycle of damage and rebuilding. FEMA estimates that mitigation saves $2-5 for every $1 spent, according to former director James L. Witt. Although some disaster relief spending is earmarked for hazard mitigation (typically no more than 10% of the total assistance), expanding this provision could be justified on the grounds of economic efficiency (Office of Technology Assessment 1993).

3. Effects of other taxes and subsidies on coastal development

3.1 State-level subsidies to industry

Although most subsidy literature focuses on federal programs, which are generally easier to quantify than state or local subsidies, Louisiana does subsidize its industrial and energy sectors at an extraordinarily high rate. Templet (1995) found Louisiana to have the highest per capita rate of perverse subsidies of all 50 states, at a level almost twice as high as the next state (Figure 2). His subsidy calculation included a composite of tax, energy, and pollution subsidies that provided economic value to industry at public expense. Though not specific to coastal development, these subsidies clearly influence the state's political and economic climate. Additionally, some percentage of these subsidies is likely relevant to key coastal industries like oil, gas, and shipping. Subsidies specific to these industries include property tax exemptions, lower energy costs as compared to residential consumers, and legacy costs of hazardous waste cleanup (Louisiana Environmental Action Network 1997). A variety of additional subsidies are offered by the Louisiana Department of Economic Development, with additional programs implemented by the state to attract business investment following Hurricane Katrina.

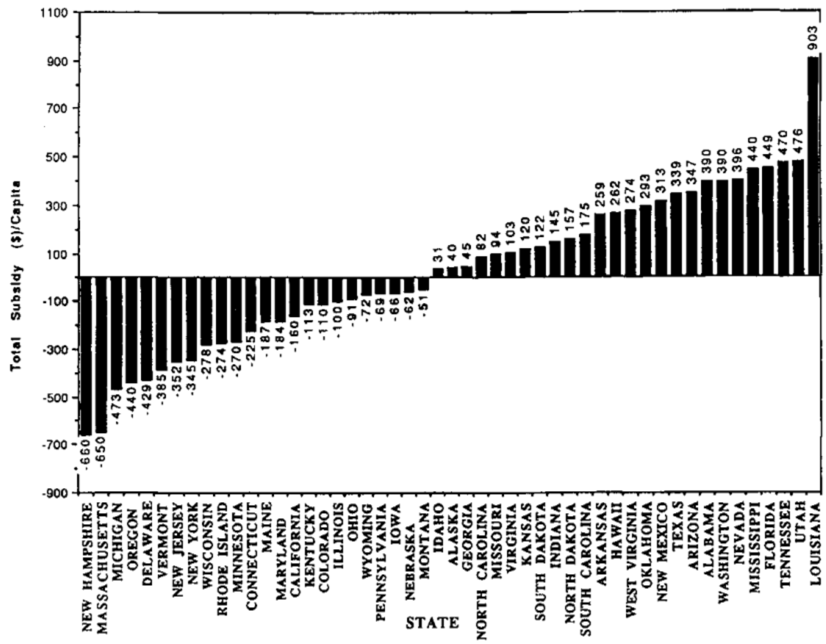

Figure 2: Total subsidy by state relative to the national average (Templet, 1995).

3.2 Federal oil and gas subsidies

Oil and gas extraction are key sectors of Louisiana's economy, creating an estimated $13.7 billion in Gross State Product in 2003 (US Bureau of Economic Analysis 2005), although production has steadily declined since the state's peak oil and gas production in 1969 and 1970, respectively (Lam 2004). By encouraging extraction of economically marginal energy sources, these tax breaks and subsidies deny a public revenue source and encourage expansion of these activities beyond sustainable scale. Most U.S. energy extraction subsidies and tax breaks were implemented in the 1970s in response to the OPEC energy crisis, with the intent of promoting development of domestic energy sources. Tax breaks include deductions for "intangible costs" of exploration, including all costs without salvage value (labor, fuels, material, power, and supplies), as well as a depletion allowance that permitted depreciation of 22-27.5% of gross income between 1926 and 1975. Today, independent producers can still claim a 15% depletion allowance for their first 1,000 barrels of oil or 6 million cubic feet of gas extracted per day (Office of Environmental Policy and Compliance 1994). This depletion allowance was estimated to be worth $260 million in 1999 (1999 dollars; Energy Information Administration 1999).

Weak enforcement of the Clean Water Act and an uneven record by the oil and gas industries at cleaning up and retiring aging wells and infrastructure are additional off-budget subsidies that distort the full cost of oil and gas extraction. Until the 1990s, minimal regulatory oversight existed for wetland loss, which was caused in part by oil and gas canal construction. By subsidizing the cost of fossil fuel extraction and use, economically inefficient overuse of these resources has taken place, leading to higher social costs (Gately 2007).

3.3 Levees, navigation, and wetlands – U.S. Army Corps of Engineers programs

The U.S. Army Corps of Engineers (Corps) is responsible for maintaining coastal Louisiana's levee and navigation systems. New Orleans' current levee system cost $12 billion to construct and maintain (LACWCRTF 1993, as cited in Cardoch and Day 2001). Prior to Katrina, four additional new levee projects were under construction at a total cost of $1.44 billion (U.S. Army Corps of Engineers 2005). For FY 2000, the Corps' New Orleans District budget was $461 million, of which $328 million went to drainage, flood control, and dredging projects, with another $47 million spent on wetland restoration in large part to mitigate for wetland impacts of the District's drainage, flood control, and dredging activities (Cardoch and Day 2001).

Outside New Orleans, $5.9 billion was spent through 1985 to build and maintain over 3,540 km of levees as part of the Mississippi River and Tributaries Project (Office of Environmental Policy and Compliance 1994). Authorized following the Great Flood of 1927, this project led to the construction of levees throughout the Lower Mississippi River, greatly reducing sediment deposition in the coastal wetland zone. The Corps has also spent $280 million to construct 800 km of navigation channels, and pays $40 million annually for their maintenance (Office of Environmental Policy and Compliance 1994). Water Resources Development legislation passed in the 1990s has shifted some of the burden for channels maintenance to the shipping and oil industries, reducing this subsidy.

Additionally, the Corps undertakes beach nourishment and provides structural protection, spending $40-70 million annually nationwide to "protect" eroding shorelines (Office of Technology Assessment 1993). These programs act as subsidies by providing free storm protection for coastal property owners. Although used sparingly in coastal Louisiana due to the coastal plain's marshy nature, beach nourishment is an important development subsidy in other U.S. coastal regions. Jones and Mangun (2001) discuss beach nourishment as a disaster mitigation strategy, but recommend integrating economic, social, and environmental analysis of costs and benefits for each project, as well as more effective public participation in the decision making process. They also recommend that beach nourishment be funded through

Disaster Relief and National Flood Insurance programs, to fulfill program mitigation requirements, provide better long-term protection, and reduce rebuilding costs.

Many Corps programs lead to impaired wetland habitat, resulting in a loss of ecosystem services, including coastal wetlands' critical flood protection function. Such wetland destruction is an off-budget cost of many Corps projects. Enforcement and mitigation requirements associated with the Clean Water Act have generally improved in recent years, although at the national level, wetland mitigation sites often fail to meet the ecological functions and values of the wetlands they purport to replace (Spieles 2005). Some destructive projects are not going forward, such as a proposed enlarging of the Intracoastal Waterway that would have eliminated 500-1,050 ha of wetlands. Additionally, freshwater and sediment diversions built by the Corps are slowing the processes of saltwater intrusion and wetland loss. The Caernarvon, Davis Pond, and Diversion to Lake Ponchartrain Basin freshwater diversions are projected to prevent the loss of 890 ha of wetlands a year, at a total cost of $173 million. The West Bay Sediment Diversion project, costing $8.5 million, will create 3,965 ha of wetland (Office of Environmental Policy and Compliance 1994). Assuming a conservative ecosystem service value of almost $15,000 per wetland ha (1994 dollars, Costanza et al. 1997), these projects provide a far better return on investment than the perverse taxes and subsidies that dot coastal Louisiana’s landscape.

Like most federal programs, the Corps has been in the process of transferring some portion of its program costs to state and local governments. This transfer of financial responsibility to the local level may reduce the number of economically marginal projects that go forward, as state and local governments more fully appreciate their costs.

3.4 Infrastructure – bridges and highways

As elsewhere in the U.S., federally subsidized bridge and highway construction has occurred in coastal Louisiana. From 1967 to 1976, 96 km of highways were constructed in the state's coastal zone, at a cost of $65.8 million. These projects also came with off-budget costs, such as the direct destruction of 760 ha of wetlands (Office of Environmental Policy and Compliance 1994). Additional construction has since taken place for Interstates 10 and 310. New publicly-funded highways also increase development pressures in areas where it might not otherwise occur. The Lake Pontchartrain Causeway, which facilitated rapid growth in St. Tammany Parish, is one such example near New Orleans.

3.5 Other infrastructure development subsidies

Various government agencies provide a range of other subsidies to build infrastructure that facilitates coastal development. These programs include: 1) Electrical system loans through the Rural Utilities Service, 2) Small business and disaster assistance loans through the Small Business Administration, 3) Community facility loans through the Farm Service Agency, 4) Various loans to business, industry, and rural housing, and 5) Housing loans through the Department of Housing and Urban Development and the Veterans Administration (Office of Technology

Assessment 1993). Loans through HUD and the VA are not available for flood zone construction in communities not certified through the NFIP, although equity and justice questions certainly arise when economically disadvantaged groups are knowingly placed in high-risk areas. It is interesting to note that all of these subsidies are prohibited within the Coastal Barrier Resources System (discussed subsequently), showing that the Federal Government clearly understands their stimulus on development.

3.6 Tax breaks to homeowners

Several tax breaks encourage residential development in coastal regions. A casualty loss deduction allows property owners to deduct the cost of uninsured damages from coastal disasters, providing a disincentive to buy flood insurance or maintain a proper level of coverage. Interest and property tax deductions are also provided for second homes, along with accelerated depreciation schedules for seasonal rental properties. Because second homes and rental properties comprise a large proportion of new coastal development, these programs provide a direct incentive for coastal development (Office of Technology Assessment 1993). These tax breaks are clearly regressive, their benefits accruing overwhelmingly to the wealthy, and the development they encourage leads to further taxpayer burden for future disaster relief.

3.7 Coastal Barrier Resources Act and Coastal Zone Management

The Coastal Barrier Resources Act (CBRA) was passed in 1982, with the goals of saving tax dollars, preventing high-risk development, and protecting ecologically valuable coastal areas. President Regan, on signing the bill, said "it will save American taxpayers millions of dollars while at the same time, taking a major step forward in the conservation of our magnificent coastal resources. (The Act) will not prohibit a property owner from building on his property… instead, it simply adopts the sensible approach that risk associated with new private development in these sensitive areas should be borne by the private sector, not underwritten by the American taxpayer" (White House 1982).

CBRA prohibits federal spending for roads, water, wastewater, or other infrastructure, and bans the purchase of flood insurance in designated areas, placing responsibility for rebuilding costs on property owners. Amendments in 1990 expanded the size of the system but allowed federal disaster relief funds to be used in these areas, contrary to the program's original intent. Today, the system includes 526,000 ha, plus an additional 728,400 ha of "Otherwise Protected Areas" where federally-subsidized infrastructure can be built but flood insurance is unavailable.

CBRA has had mixed success in achieving its goals. It is projected to save taxpayers $1.3 billion (1996 dollars) from 1983 through 2010 in avoided infrastructure costs, insurance, and disaster relief (U.S. Fish and Wildlife Service 2002). Although USFWS (2002) did not break down cost savings by state, Louisiana saved over $1 million in Disaster Relief between 1988 and 1996 (1996 dollars; although this is a small number compared to other dollar figures in this paper, this may be due to relatively sparse settlement of Louisiana CBRA units). Most CBRA units have not been developed, although where pressures to build are high and state and local development subsidies exist, CBRA alone will not protect valuable coastal areas (Salvesen 2005). Examples include Bethany Beach, Delaware, and Cape San Blas, Florida, which developed similarly to adjacent unprotected areas. In another case, road and bridge repairs to a densely-populated CBRA unit, Topsail Island, North Carolina, were rebuilt at public expense following Hurricane Fran in 1996 (Platt et al. 2002).

State and local governments and conservation groups can work with CBRA to further strengthen it, providing for land acquisition funding and limiting local subsidies. Connecticut, Florida, Maine, Massachusetts, Rhode Island, and Texas have laws limiting coastal development subsidies, although these programs differ in their enforcement and in specific prohibited subsidies (Godschalk et al. 2000). Unfortunately, states also sometimes work against CBRA's goals, providing their own subsidies that facilitate a thriving coastal development industry (Salvesen 2005). Berke (1999) suggests that a mix of incentive and regulatory programs at the state and local level will be most effective in promoting development that is sensitive to disaster risk, but perhaps the best solution is to transfer the risk of coastal development to residents (by limiting federally-subsidized insurance) and developers (by limiting subsidized infrastructure).

Another federal program, Coastal Zone Management (CZM), provides states a framework and funding to manage their coastal zones. CZM is an example of a program that can supplement or work against CBRA. State CZM programs are difficult to compare, but Louisiana's places a high priority on coastal wetlands and estuaries and seaports, and a low priority on public access, urban waterfronts, beaches, bluffs, and rocky shores (Hershman 1999, Good et al. 1999, Bernd-Cohen and Gordon 1999, Goodwin 1999, Pogue and Lee 1999). Of particular relevance to wetlands, these studies rated Louisiana's CZM program as moderately effective (versus model CZM policies). Louisiana's wetland management scored highly, since the state is moving forward with voluntary wetland protection and restoration measures. Also relevant is the fact that dredge spoil from seaport maintenance must be used for ecological benefit, such as wetland restoration. Unfortunately, although Louisiana's CZM program received high scores, the state still faces unique and daunting challenges in protecting its coastal zone.

3.8 Fragmentation of federal programs

A survey of federal programs influencing coastal development and disaster relief shows a generally piecemeal approach (Table 2). Those programs that do have explicit goals of limiting risky development, protecting natural resources, and reducing government and taxpayer liability to pay for repeated reconstruction are badly undermined by a range of programs that subsidize such development (Office of Technology Assessment 1993).

Birkland (2001), in comparing national hurricane and earthquake policy, emphasizes the ineffectiveness of the U.S.'s fractured approach to hurricane response.

Seismic engineers and others successfully advocated for the National Earthquake Hazard Reduction Act (NEHRA), which has integrated planning, development, and disaster response along known earthquake fault lines. The 2005 hurricane season demonstrated the potential value of an integrated National Hurricane Hazard Reduction Act (NHHRA), as suggested by Birkland, that would sensibly direct future coastal settlement patterns, while minimizing current and future taxpayer burdens and transferring the risks of coastal development to those who choose to live in high-risk areas. Alternatively, improved standards and enforcement of current policies might be deemed preferable to creation of a new policy layer.

Table 2: Tax, subsidy, and insurance policies relevant to coastal development and disaster relief

Category	Program	Original intent	Financial status	Effects on development
Insurance	National Flood Insurance Program	Discourage flood zone development; reduce disaster relief payments.	Intended to be self financing, but must borrow following major storms. Serious problems with repetitive loss; reforms in 2004 and 2006 may help somewhat.	Subsidizes risky development, especially repetitive loss property owners, developers, private insurers. Problems with information asymmetry, moral hazard.
Disaster relief	Stafford Disaster Relief	Assist disaster victims.	Average of $3 billion per year spent from 1990-2003; costs likely to rise with coastal population growth and climate change.	Entitlement program for coastal property owners; provides financial incentive for otherwise marginally profitable development.
Mitigation/ Relocation Assistance	Flood Mitigation Assistance	Relocate victims, encourage spending to reduce future losses.	Estimated by FEMA to save $2-5 for every dollar spent.	Relocation has seen limited use in coastal Louisiana; could use to provide large savings in future if used.
Industry subsidies and tax breaks	State subsidies and tax breaks to industry	Provide benefits to local industry and business.	Louisiana has largest industrial subsidy per capita of any U.S. state; weakens state finances.	Encouraged industrial development including oil and gas industries in coastal wetlands.
	Oil and Gas Subsidies	Encourage domestic energy production during OPEC crisis.	No dollar value found for deductions for exploration; depletion allowance worth $260 million in 1999.	Encouraged extraction of marginal energy sources; increased canal development and wetland loss.
U.S. Army Corps of Engineers activities	Levees, navigation, and shoreline protection	Coastal storm protection; maintain shipping industry.	$12 billion for New Orleans' levee system; $1.4 billion for other levees; $5.9 billion for Mississippi River and Tributaries Project; $280 million plus $40 million annually for navigation channels; $40-70 million annually (nationwide) for beach nourishment and shoreline protection.	Facilitated development in New Orleans, led to direct and indirect wetland destruction. Navigation channel maintenance a direct subsidy to shipping, oil, and gas industries.
	Wetland restoration programs	Reduce saltwater intrusion and wetlands loss in Coastal Louisiana.	$182 million for four freshwater and sediment diversion projects.	Thousands of hectares of wetlands created or preserved; value of protected and restored ecosystem services likely exceeds cost of projects.
Infrastructure subsidies	Highway construction	Government spending on built capital.	$66 million for 96 km of highways in coastal Louisiana.	Direct destruction of wetlands; facilitated sprawling coastal development.

	Miscellaneous infrastructure subsidies and loan programs	Provide built capital and loans to commerce, industry, and residents.	No dollar value found; subsidies for electrification, bridge construction, loans to homeowners, business owners, industry, housing programs through HUD and VA.	Encouraged new development by providing financial incentive for otherwise marginally profitable development.
Tax	Homeowner tax breaks	Depreciation for uninsured disaster damage; deductions on second homes and rental properties.	No dollar value found.	Directly encourage otherwise marginal, risky construction. Regressive tax since benefits accrue to second homeowners and rental property owners.
Market based mechanism	Coastal Barrier Resources Act	Remove subsidies to coastal development in designated zones.	Estimated to have saved taxpayers $1.3 billion over 27-year period.	Has slowed development and limited taxpayer spending on disaster relief in some areas, but does not stop all coastal development.

4. The Gulf Coast's future: Encouraging desirable redevelopment

4.1 Initial planning for post-Katrina reconstruction: September 2005-January 2006

Tax, subsidy, and insurance reform issues have all been part of the proposed reconstruction plans discussed following Katrina. Initial proposals by the Bush administration included a "Gulf opportunity zone," providing tax incentives to businesses relocating in storm-damaged areas and "Urban homesteading" to provide low-income residents with federally-owned land to rebuild housing. These proposals made some provision to address the income distribution gaps that Katrina revealed (providing low-income housing and support for minority-owned and small businesses), but also would likely subsidize industries providing questionable social benefits (such as casinos).

Conservative think tanks such as The Heritage Foundation produced their own policy proposals that they purport will speed the recovery of the Gulf Coast (Meese et al. 2005). Meese et al. fail to distinguish between perverse subsidies that are economically inefficient and socially or environmentally destructive, versus beneficial subsidies that can produce socially desirable outcomes in excess of their costs, increasing both efficiency and society's well-being. These proposals also fail to address allocation, distribution, and scale questions for the Gulf Coast. Farley et al. (2007) provide a full discussion of "market fundamentalist" proposals to rebuild New Orleans.

Amidst this wave of initial policy proposals, Louisiana's congressional delegation produced the Blueprint for Federal Response to Hurricane Katrina and Rita (S.1765 2005). This controversial bill requested $250 billion in federal aid, earmarked for a variety of rebuilding efforts addressing built, human, social, and natural capital. It proposed a wide range of subsidies to individuals and businesses, some of which are likely perverse subsidies. Although a full evaluation of the allocation, distribution, and scale attributes of the Blueprint's provisions is beyond the scale of this paper, sound proposals to rebuild natural capital are part of the Blueprint. It proposes about $10 billion to the U.S. EPA and Louisiana Department of Environmental Quality to address hazardous waste, air and water pollution, and water treatment, and for restoration of Lake Pontchartrain. It also recommends $40 billion for hurricane protection, flood control, coastal wetland restoration, and navigation activities (activities typically within the domain of the Corps). The Blueprint also proposes to expand Louisiana's state boundary seaward onto the Outer Continental Shelf, which would allow the state to capture about $2.5 billion per year in new rents from oil and gas extraction, which it would spend on further coastal restoration and hurricane protection. This portion of the Blueprint recently received support in the form of the controversial Gulf Coast Protection Act, which proposed to increase offshore oil and gas drilling in the Gulf but return royalties to the states to fund wetland restoration and structural flood protection projects.

Taken together, these proposals could provide funding for Coast 2050 (the state's coastal wetland restoration plan, discussed subsequently in further detail), while using rents captured from nonrenewable resource extraction to protect the state's natural and built capital. These appear to be sound steps to improve the Gulf Coast long-term sustainability and economic well-being. We propose several such additional policy solutions addressing allocation, distribution, and scale in the following sections.

4.2 Toward unified, efficient, just, and sustainable policy solutions for New Orleans

Ecological economics seeks policy solutions leading toward an efficient allocation of goods and services by market and nonmarket systems; a just distribution of resources in today's society and for future generations, and a sustainable macroeconomic scale that does not undermine the resources of future generations and the existence of nonhuman species that provide critical ecosystem services (Daly 1992, Farley et al., 2007). Current policy tools differ in their impacts on allocation, distribution, and scale; those with negative effects on all three represent perverse subsidies worthy of elimination (Table 3). Policies with some desirable effects may require reform, but be worth continuing. Additionally, new policy instruments should be considered to promote a healthy local economy while preserving and restoring natural, social, human, and built capital.

Part of the problem with the existing group of policies is that they were implemented at different times, and with widely varying goals, some of which are no longer appropriate. New policies should provide for the needs of coastal residents while promoting efficient allocation, just distribution, and sustainable scale. We list such policies (Table 4) as a synthesis of our proposals and those of other authors viewed through these ecological economic criteria. Many have been previously suggested by others (Office of Technology Assessment 1993, Godschalk et al. 1999, Godschalk et al. 2000, Platt et al. 2002); we also evaluate their proposals through the lens of ecological economics. Note that these proposals are independent of those included in the above-discussed Blueprint.

Table 3: Effects of existing government policies on allocation, distribution, and scale

Program	Allocation effects	Distribution effects	Scale effects
National Flood Insurance Program*	NEGATIVE. Information asymmetry, moral hazard, repetitive loss result in inefficient program.	NEGATIVE. Small minority of policyholders receive most payout; subsidy also provided to private insurers who sell through NFIP.	NEGATIVE. Provides incentive for development in sensitive floodplains and coastal zones, leading to loss of natural capital.
Stafford Disaster Relief*	NEGATIVE. Full social cost of living in disaster prone regions not represented in the market.	NEGATIVE. Full costs paid by public subsidy to those living in high-risk zones.	NEGATIVE. Provides incentive for development in disaster prone areas, leading to loss of natural capital.
Mitigation and Relocation Assistance	NEUTRAL to POSITIVE. Can result in more efficient housing patterns and lower subsidies to homeowners in high risk locations.	POSITIVE. Can minimize future costs to taxpayers for disaster relief, reconstruction costs.	NEUTRAL to POSITIVE. Negative if relocation leads to construction in sensitive areas; positive when purchased land is left as public open space.
State and federal oil and gas subsidies*	NEGATIVE. Distorts prices, so full costs are not accounted for in oil/gas production or purchase price.	NEGATIVE. Transfers wealth from public to oil and gas industries.	NEGATIVE. Encourages destruction of natural capital (wetlands); discourages conservation, renewable energy, and sustainable energy policy.
Levees, navigation, shoreline protection*	NEGATIVE. Distorts costs for oil, gas, shipping, and coastal development.	NEGATIVE. Transfers wealth to navigation, oil, gas, and coastal development industries.	NEGATIVE. Destroys natural capital.
Wetland restoration	POSITIVE. Economic benefits of ecosystem services usually outweigh costs.	POSITIVE. Society and future generations benefit from ecosystem services.	POSITIVE. Preserves natural capital.
Highway Construction*	NEGATIVE. Subsidies encourage construction under marginal economic conditions.	NEGATIVE. Costs, particularly during reconstruction, are public subsidy.	NEGATIVE. Destroys natural capital directly through construction and indirectly, facilitating more new development.
Miscellaneous infrastructure subsidies*	NEGATIVE. Artificially lowers costs to coastal developers and landowners, distorting costs and incentives.	NEGATIVE. Transfers wealth disproportionately to developers. (HUD, VA loans may be POSITIVE, assisting economically disadvantaged with homeownership).	NEGATIVE. Destroys natural capital, encourages poorly-planned coastal development.
Homeowner tax breaks*	NEGATIVE. Distorts full cost to live in high-risk areas.	NEGATIVE. Transfer to wealthy (owners of second homes and rental properties).	NEGATIVE. Encourages development of coastal areas and loss of natural capital.

Coastal Barrier Resources Act	POSITIVE. Removes subsidies to better show full cost of coastal development.	POSITIVE. Does not require taxpayers to repeatedly pay for reconstruction of damaged areas.	NEUTRAL to POSITIVE. Slowed but not stopped development in sensitive areas. Local, state policies may improve or reduce effectiveness.

* Defined as a perverse subsidy – economically inefficient while damaging natural and/or social capital

Table 4: Effects of proposed government policies on allocation, distribution, and scale

Program	Allocation effects	Distribution effects	Scale effects
Phase out NFIP	POSITIVE. Reduce incentive for flood zone development and habitation.	POSITIVE. Reduce subsidy to highest-risk property owners.	POSITIVE. Discourage inappropriate coastal and floodplain development while potentially allowing construction if developer/owner bears burden.
Disaster relief reform	POSITIVE. Reduce incentive for flood zone development and habitation.	POSITIVE. Reduce subsidy from those living outside flood zones to those in high-risk zones.	POSITIVE. Discourage new high-risk development potentially allowing construction if developer/owner bears burden.
Expand use of mitigation and relocation assistance	POSITIVE. Reduce future subsidies to residents of flood zones.	POSITIVE. One-time costs reduce future public disaster aid burden.	Generally POSITIVE, especially if reclaimed land is used for public open space and new construction does not deplete natural and social capital.
End perverse subsidies on oil and gas extraction, institute extraction tax	POSITIVE. Improve accuracy of economic incentives, leading to extraction only from economically viable energy sources.	POSITIVE. Eliminate transfer of wealth to oil companies.	POSITIVE. By increasing price of energy, would encourage conservation and use of renewable energy.
Financial assurance bonds for oil, gas, and navigation canals	POSITIVE. Require industry to pay full social costs, allowing use and cleanup if economically efficient.	POSITIVE. End pattern of industry passing restoration costs on to public.	POSITIVE. Prevent future loss of natural capital from canals.
Fund Coast 2050 Program	POSITIVE. Maintain flow of ecosystem services and underprovided public goods.	POSITIVE. Minimize loss of wetlands for future generations.	POSITIVE. Preserves and restores critical natural capital.
Eliminate perverse subsidies on new coastal infrastructure	POSITIVE. End incentives for inefficient development, leading to more efficient market.	POSITIVE. Reduce transfer of wealth to land developers and landowners.	POSITIVE. Reduce rate of consumption of natural capital.
Reform tax breaks for homeowners	POSITIVE. Encourage spending on mitigation measures to reduce future reconstruction costs.	POSITIVE. Reduce tax breaks to generally wealthy second homeowners.	POSITIVE. Discourage second home construction; encourage use of mitigation measures on existing construction.
Strengthen Coastal Barrier Resources Act	POSITIVE. Eliminate artificial incentives to reflect full development costs, lead to more efficient market.	POSITIVE. Reduce transfers of wealth from public to flood zone residents.	POSITIVE. Preserve coastal natural capital and discourage inappropriate development.
Tax land in coastal cities and states	POSITIVE. Tax on an inelastically supplied good is efficient, resulting in zero deadweight loss.	POSITIVE. Progressive tax that shifts burden generally to large landowners.	POSITIVE. Encourages high-value uses of land, reducing underutilized urban land, incentive for land speculation, and natural capital loss.

4.3 Eliminate perverse subsidies

Many existing perverse subsidies should be eliminated. Tax breaks to the oil and gas industries, to homeowners of coastal developments, and for new publicly-funded infrastructure in coastal zones fall into this category. These programs are economically inefficient, environmentally and/or socially damaging, and benefit the few and often wealthy and politically well-connected at the expense of the vast majority of U.S. taxpayers. Advocates of subsidies often argue that cutting industry subsidies will hurt employees of these companies, but Templet (1995) showed that there is no relationship between subsidy rate and employment level, and that reducing state-level subsidies in fact led to greater employment in Louisiana.

Like most perverse subsidies, the benefits of publicly-funded coastal zone built capital (roads, electricity, water and wastewater) accrue to a small group of individuals. An appropriate market-based solution would require developers to pay the full costs of infrastructure provision to all areas. In this way development in profitable areas lacking high ecological or social value might still occur, but those in economically marginal areas would not be built (CBRA's central premise). Such policy shifts the economic burden from the public to those who benefit from development. Another example is the Corps' maintenance of shipping channels through Louisiana's coastal wetlands, to the benefit of the navigation, oil, and gas industries. Recently the Corps has begun requiring industry to share these maintenance costs. This is a good start, but ideally the full cost should be transferred to the party benefiting from the subsidy, allowing economically viable activities to continue while economically inefficient uses are phased out.

Disaster relief and the NFIP, as currently administered, function as perverse subsidies. Although few would argue against some form of government assistance after a disaster the size of Hurricane Katrina, the current system encourages rebuilding the same infrastructure to be knocked down by the next storm. Reforms to disaster relief could include raising state and local cost sharing (at least back to the original 25% program requirement), increased spending on mitigation, and tightening the criteria for presidential disaster declarations.

Given the NFIP's failure to control flood damage losses, reconsidering its role as the nation's primary flood policy tool would be valuable. Ideally the government would exit the insurance business – an inefficient program that has not successfully reduced the nation's flood risk (Cummins 2006) – and allow those who could afford to privately insure their property to do so. Cummins also suggests that new financial tools such as catastrophe bonds may make private flood insurance programs more feasible. Although eliminating the NFIP would be politically challenging, a gradual phase-out or additional reform might be more feasible. An extended relocation assistance program could ease the burden on those who could not afford private insurance. In the interim, the NFIP should work to minimize the information asymmetry (Chivers and Flores 2002) and moral hazard problems inherent in the program. California's 1998 Natural Hazard Disclosure Law appears to have improved insurance-related information asymmetry problems in that state, and could serve as a model policy (Troy and Romm

2004). Local governments can also work with the NFIP to develop more stringent elevation standards for flood zone construction (Holloway and Burby 1990) through the NFIP's Community Rating System. Finally, premiums should be increased to reflect risk, and coverage should be eliminated for repetitive loss properties. The 2004 Flood Insurance Reform Act took a first step toward addressing repetitive loss, and the proposed 2006 Flood Insurance Reform and Modernization Act takes steps toward charging actuarially sound rates for some properties.

The Flood Mitigation Assistance relocation program could be more widely implemented in hurricane-prone areas. Past relocation programs have had only marginal success, since flood zone residents typically have economic incentives to rebuild versus relocate. However, removing the perverse subsidies that help finance flood zone residency could improve participation. Although such programs have been criticized as giveaways to property owners, they actually save money by eliminating the cycle of repetitive loss properties (Godschalk et al. 1999). Pilkey and Young (2005) advocate an organized retreat from the highest-risk coastal areas. This process can also be used to acquire open space and restore natural capital. Relocation programs should be sensitive to the fact that relocation and natural disasters can have complex effects on social capital. They should also avoid damaging natural capital through poorly planned development. Interestingly, relocation has been proposed for traditionally marginalized communities such as New Orleans' Lower Ninth Ward and Cameron Parish in western Louisiana (Longman 2005), but not for wealthier Gulf Coast communities.

4.4 Reform existing programs

Some programs improve economic efficiency, allocation, and/or scale, but are poorly managed or are failing to fully achieve their goals. Such programs, like CBRA, would benefit from reform. Although many CBRA units have remained undeveloped, most have limited access or are dominated by wetlands, making them inherently difficult to develop. CBRA could be extended to new areas while reducing the few subsidies still available within units (i.e., homeowner tax breaks, disaster relief). A loophole for road reconstruction subsidies should also be removed from CBRA – in the past, developers have designated roads for public use, and by virtue of their connection to the nation's road network, received federal funding for reconstruction following a disaster (Salvesen 2005).

Probably the biggest obstacle to CBRA's success is the uneven approach to coastal management taken by state and local governments. Improving state and local initiatives to work with, and not against CBRA could take many forms, as suggested by Berke (1999). States could be required to address CBRA goals as part of their federally-approved CZM plans, replacing perverse subsidies with those that improve public welfare. Finally, the U.S. Fish and Wildlife Service has enforcement authority for CBRA, but has little power to enforce the Act, and has been overruled on development decisions in the past (Godschalk et al. 2000). Increasing the authority of this agency or delegating enforcement to another agency might improve the Act's effectiveness.

4.5 Consider new coastal policies

Several new policies could add stability to U.S. coastal regions by providing economic development incentives to reduce risk while preserving natural capital. Homeowner tax breaks are a logical starting point. For homeowners still living in flood zones, the current perverse subsidies that encourage development and discourage NFIP participation should be replaced with tax incentives for homeowner spending on mitigation, which reduces the risk of future flood damage.

Further loss of Louisiana's coastal wetlands should be strongly discouraged. Performance bonds (Costanza and Perrings 1990, Office of Environmental Policy and Compliance 1994) are one tool that allows temporary use of an area for resource extraction. Performance bonds require the extractive industry (i.e., oil or gas company) to post a bond of value equivalent to the operation's full clean-up and restoration cost. This way, the burden of responsibility falls on industry rather than the public. While industry can still operate, it has an economic incentive to minimize environmental damage.

Additionally, the Coast 2050 wetland restoration plan should be considered for full funding. First proposed in 1998, Coast 2050's estimated $14 billion price tag was deemed too expensive; Louisiana in particular could not fund its required state cost share. Although scientific consensus is that additional coastal wetlands would not have protected New Orleans proper from flooding due to Hurricane Katrina's path (National Academy of Sciences 2006), a viable coastal wetland system would provide flood protection from future storms. Louisiana's coastal wetlands also sustain the largest fisheries of any U.S. state outside Alaska, a major local employer, and provide numerous other ecosystem services (Costanza et al. 1989).

Finally, shifting taxation to land offers the opportunity to discourage inefficient land development patterns and encourage use of underutilized urban land, while using an economically efficient, progressive tax (Daly and Farley 2004). The tax base is best shifted from property to land gradually, in conjunction with community outreach to allow local adjustment to new incentives (Hartzok 1997). Taxing land in Louisiana would be a way to encourage redevelopment of New Orleans while limiting sprawling development into high-risk, vulnerable coastal areas.

5. Conclusion

Inconsistencies in data reporting make it impossible to obtain a total dollar value for the various subsidies to coastal development, especially for coastal Louisiana. Put in light of the estimated $200 billion reconstruction bill for Hurricane Katrina, there is clear value in reconsidering the current system of taxes, subsidies, and insurance in high-risk coastal zones. Katrina highlighted problems with current U.S. coastal policy – one that encourages an unending cycle of risky development and massive disaster relief – and the desirability of a policy that is unified, efficient, just, and sustainable. Such thorough reconsideration of coastal management policies could help move toward improved fiscal responsibility and increased preparedness for future disasters so that loss of life and reconstruction costs are smaller the next time a major storm rolls across the Gulf Coast.

Acknowledgements

We thank Melissa Bailey, Joshua Farley, Brian Miles, Brian Voigt, and one anonymous referee for valuable review comments on earlier versions of this manuscript.

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CHAPTER 3: OPPORTUNITIES AND CHALLENGES IN APPLYING THE GENUINE PROGRESS INDICATOR/INDEX OF SUSTAINABLE ECONOMIC WELFARE AT LOCAL SCALES[1]

Abstract

The closely related Genuine Progress Indicator (GPI) and Index of Sustainable Economic Welfare (ISEW) provide monetized estimates of societal well-being based on economic, social, and environmental criteria. Although the first ISEW/GPI estimates were completed at the national scale, there has been recent interest in applying GPI locally and regionally. Similar to national policy decisions, local fiscal, environmental, and land use choices can strongly influence well-being. Local GPI estimates present several challenges, including data quality and availability, interpretation of certain components, and appropriate application of results. We present a case study from seven counties in northern Vermont, USA from 1950-2000. This case study facilitates comparison between county, state and national GPI, and across a small urban-rural gradient. The case study illustrates both the difficulties and value of applying GPI/ISEW at local scales. We find that for recent years in an industrialized nation, it is possible to construct robust GPI estimates that allow comparisons of well-being across regions.

[1] Bagstad, K.J. and M. Ceroni. 2007. Opportunities and challenges in applying the Genuine Progress Indicator/Index of Sustainable Economic Welfare at local scales. International Journal of Environment, Workplace, and Employment 3 (2): 132-153.

Keywords

Genuine Progress Indicator, Index of Sustainable Economic Welfare, urban-rural, social welfare, quality of life, regional development

1. Introduction

Recent years have witnessed a growing interest in developing meaningful quality of life measures at the community level, with the implicit goal of improving quality of life (Haggerty et al. 2001). Gross domestic product (GDP), despite its many flaws, is still frequently used as a proxy measure for society's welfare, despite a growing body of literature suggesting that increased wealth and economic output alone do not always improve quality of life or subjective well-being for individuals or society (Cobb et al. 1995, UNDP 1996, Diener et al. 1999, Eckersley 2000, Frey and Stutzer 2002, Kahneman 2004, Easterlin 2005). Economists and politicians, including some of the original architects of GDP accounting have also noted the misuse of GDP as a welfare measure.

Since at least the late 1960s, economists have attempted to adjust GDP to better measure society's well-being. Early efforts to incorporate social and environmental costs into GDP included those of Sametz (1968), Nordhaus and Tobin (1972), and Zolotas (1981). In perhaps the most well-known of these studies, Nordhaus and Tobin concluded that as of the early 1970s, economic growth was leading to improvements in quality of life in the United States. Daly and Cobb (1989) revisited Nordhaus and Tobin's findings with their Index of Sustainable Economic Welfare (ISEW), which was later revised as the Genuine Progress Indicator (GPI). The ISEW/GPI (hereafter referred to as GPI) begins with a measure of personal consumption, weighted to account for income inequality, and deducts or adds value for various monetized measures of built, human, social, and natural capital. This can be expressed in the form of the equation (adapted from Hanley et al. 1999):

$$GPI = C_{adj} + G + W - D - S - E - N \quad (1)$$

Where: C_{adj} = personal consumption adjusted to account for income distribution, G = growth in capital and net change in international position, W = non-monetary contributions to welfare (e.g., household labor, volunteer work), D = defensive private expenditures, S = depletion of social capital (e.g., cost of crime, family breakdown, lost leisure time), E = costs of environmental degradation, and N = depletion of natural capital.

The inclusion of these components makes GPI better suited than GDP to addressing questions of distribution, societal well-being, and sustainability within the economy. Daly and Cobb and subsequent authors found that welfare as measured by the GPI grew, though not as quickly as GDP, until the mid-1970s, and has since leveled off or declined slightly. These results agreed with Max-Neef's (1995) "threshold hypothesis", which states that economic growth improves quality of life up to a point, but eventually erodes environmental and social quality, reducing quality of life.

Studies in numerous other nations corroborated these findings (Jackson and Stymne 1996).

In evaluating the GPI, past authors have raised questions about the theory, data quality, and methods used to calculate GPI. These issues are also relevant for local to regional scale GPI calculations. Haggerty et al. (2001) evaluate 22 quality of life indices including the GPI, against 14 criteria, including public policy relevance, strength of theoretical foundation, and data availability and quality. Many other quality of life indices focus strongly on human health or economic domains or subjective happiness or life satisfaction surveys. GPI broadly includes built, human, social, and natural capital, and is one of few measures to aggregate its criteria into a monetary value. It does not however include any subjective measures of well-being, like some other quality of life measures. Like others, Haggerty et al. (2001) criticize the reliability, validity, and sensitivity of the economic methods used to estimate the value of certain component attributes of the GPI.

Several components of GPI's theoretical framework have been questioned (Neumayer 1999, Neumayer 2000, Haggerty et al. 2001). Part of this controversy relates to what GPI is intended to measure – is it an indicator of sustainable (Hicksian) income, a pure replacement for GDP, an index of economic welfare, or an assessment of how well human needs are met? Costanza et al. (2001) place national accounting systems, including GDP, various forms of "Green GDP", GPI, and other indices within a framework of environmental accounting and ability to measure the economy. Starting with GDP as a purely production and consumption-based approach, Net Domestic

Product (NDP) and Green GDP attempt to better measure sustainable income. Nordhaus and Tobin's MEW, ISEW, and GPI provide more advanced accounting for other aspects of well-being, but do not address fulfillment of human needs. Measures of subjective well-being are subject of considerable recent research interest (Diener et al. 1999, Kahneman 2004, Veenhoven 2004, Vermuri and Costanza 2006) and represent another frontier in quality of life measurement.

Hanley et al. (1999), in comparing different macroeconomic sustainability indicators for Scotland, note that different measures will give different messages about whether the economy is moving in a sustainable direction. Lawn (2003) provides a theoretical basis for using GPI as an improved measure of welfare over GDP, NDP, or Green GDP. Lawn uses Fisher's definition of income as the utility or satisfaction consumers get from the economy, as opposed to Hicks' definition of income as the maximum a household or nation can consume without reducing its ability to do so in the future. Fisher's definition of income provides a more holistic view of the economy and fits well within the paradigm of ecological economics, a rapidly growing transdisciplinary field that studies the strong interrelationships between natural ecosystems, economic systems (Costanza 1989), and quality of life (Max-Neef 1995, Costanza et al. 2007).

Like nations, sub-national political jurisdictions of all sizes are increasingly interested in measuring quality of life, and in developing policies to support social well-being. At least in industrialized nations, these economies are open and often small. Many fiscal, social, and environmental policy choices take place at the national level, and greatly affect well-being at local scales. Yet local jurisdictions also make similarly important decisions. For example, a state or province can use tax policy to encourage or discourage employment in certain economic sectors. A county or municipality can make land use or resource extraction decisions that liquidate natural capital in favor of sometimes short-term employment gains. In other cases, local jurisdictions may choose to adopt more stringent environmental or social goals than the federal mandate. Since GPI aggregates a broad suite of economic, social, and environmental indicators, it can be used as a tool, for example, to compare well-being in between two or more regions with different policies. GPI's ability to aggregate an otherwise diverse set of indicators is an important strength as a measure of well-being.

In this study, we evaluate the use of GPI at local scales. In section 2, we review past local and regional estimates of the GPI. In section 3, we develop our own local estimate of the GPI for seven northern Vermont counties from the years 1950-2000, and use this case study to explore differences in well-being across the urban-rural gradient and as compared to non-monetary assessments of well-being. Section 4 identifies methods needed to construct rigorous GPI estimates, and the benefits that these estimates can provide in measuring local and regional well-being. In section 5, we describe the challenges in developing current, accurate, and theoretically sound local GPI estimates. Section 6 presents conclusions.

2. Past local and regional scale GPI studies

The costs and benefits of economic growth are not distributed evenly across a nation. Certain regions may maintain their social capital, pursue stronger environmental protection, have a more even income distribution, or import pollution-intensive manufactured goods or energy from elsewhere. Understanding these local and regional differences has fueled recent interest in developing local-scale GPI studies. In the U.S., GPI was recently calculated for nine counties in the San Francisco Bay area (Venetoulis and Cobb 2004) and at the state, county, and city level for Vermont (Costanza et al. 2004). These studies found GPI to be consistently higher in these areas than the national average. This may be due to efforts by Vermont and the San Francisco Bay area to develop strong local economies while preserving environmental quality and social cohesion.

Outside the U.S., local GPI and ISEW studies have been conducted for Victoria, Australia (Lawn and Clarke 2006), several Canadian provinces, four Chinese cities (Wen et al. 2007), Siena, Italy (Pulselli et al. 2006), and several regions within the U.K. (Moffatt and Wilson 1994, Matthews et al. 2003, Jackson et al. 2006). These local studies do not always provide comparisons to national-level figures, often due to differing data or methods. Pulselli et al., however, note that Siena has relatively less pollution, lower population density, and more tourist and agricultural centered economy than the rest of Italy. Lawn and Clarke (2006) found higher GPI per capita in Victoria versus the rest of Australia, based on better employment opportunities in Victoria as well as better performance on some environmental indicators. Jackson et al. (2006)

found the Yorkshire and Humber region's ISEW to be growing at a slightly slower rate than the U.K., and also to lag behind the larger Northern Way region, largely due to Yorkshire and Humber's greater industrial base and costs of pollution.

These studies also revealed limitations with using GPI at local or regional scales. At least for industrialized nations, data for most of the components used in GPI are readily available at the national level. However, national statistics agencies were not originally designed to collect needed GPI-related data at local scales. In the U.S., historical data may be available only for decennial census years, leaving wide year-to-year gaps. Worse, state and local data may simply not exist. In such cases, analysts typically scale down national or state values based on variables such as population or land area. While this method provides "filler" estimates, it also obscures the local differences that are important to local quality of life, the main justification for undertaking these studies. To develop truly useful and comparable local studies, consistent methods should be developed and pertinent data gaps identified, with improved measurement, survey, and accounting methods to better assess local well-being. GPI Atlantic, a Canadian NGO, has been a leader in developing surveys to collect local data on GPI-relevant quality of life attributes for Canada's maritime provinces.

3. Case Study: GPI estimates for Vermont's Northern Forest region

3.1 Study area and methods

The U.S. Northern Forest ecoregion encompasses over 100,000 km^2 across 27 counties in northern Maine, New Hampshire, New York, and Vermont. Organizations including the Northern Forest Alliance and Northern Forest Center are working to build sustainable local economies while protecting and restoring the region's natural setting. Part of this vision includes a future where "the traditional patterns of land ownership and use are maintained to provide future generations with the same benefits we enjoy today" (Northern Forest Lands Council 1994). Using conventional economic measures, the Northern Forest contributes $19.5 U.S. billion annually to the region's economy through forest-based manufacturing, tourism, and recreation (North East State Foresters Association 2004).

In Vermont, six counties – Caledonia, Essex, Franklin, Lamoille, Orleans, and Washington are included in the Northern Forest (Figure 1). These counties are characterized by low population density, abundant forest cover, and a settlement pattern of small New England town centers. The Northern Forest economy was traditionally centered around farming, forestry, and production of forest products. Tourism and outdoor recreation have become increasingly important in recent years, and many Northern Forest towns today are working to improve local employment opportunities while preserving the region's environmental and cultural character. The three easternmost counties of Caledonia, Essex, and Orleans constitute a rather homogeneous and geographically isolated area in Northeastern Vermont that is known as the

Northeast Kingdom. Intense efforts are ongoing to brand this area as a tourist destination. By contrast, Chittenden County, the subject of a prior local GPI study, is relatively urban by Vermont and Northern Forest standards (Table 1). Chittenden County is Vermont's most populous county. It includes the state's largest city, Burlington, as well as the largest employers in the state.

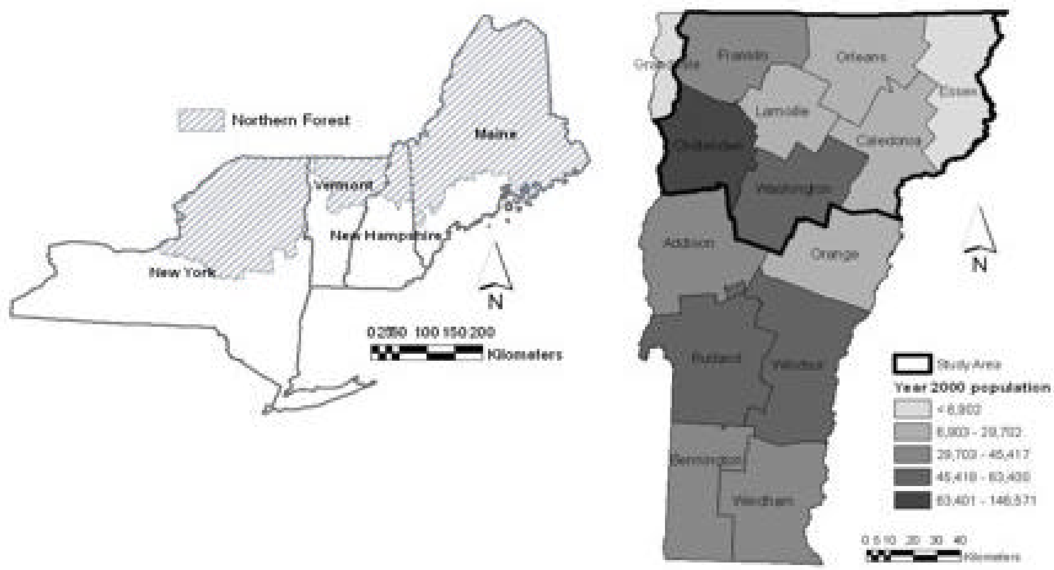

Figure 1: Northern Forest boundary and Vermont counties within the Northern Forest

Table 1: Comparison between rural and urban counties in Northern Vermont

	Six Northern Forest counties	Chittenden County
Population density, 2000 census (persons/km^2)	19.2	105.0
Percent forested	81%	61%
Largest city population	9,291 (Barre)	38,889 (Burlington)

To calculate the GPI for the Northern Forest counties, we strove to maintain consistency by following the methods of Costanza et al. (2004), who in turn followed those of Anielski and Rowe (1999). Since Anielski and Rowe's calculations were at the national level, Costanza et al's local-level adjustments were used as appropriate. When improved data sources or methods were available, we noted these changes. Following Costanza et al., we calculate values for the decennial years 1950-2000, for the 26 components of GPI for the six Northern Forest counties in Vermont (Caledonia, Essex, Franklin, Lamoille, Orleans, and Washington). Where there were changes in methods or data sources, we also recalculated values for Chittenden County to allow comparison between a relatively urban county (Chittenden) to more rural counties of northern Vermont. As such we discuss GPI results from seven of Vermont's 14 counties. All monetary values were converted into year 2000 U.S. dollars using the Consumer Price Index (CPI) from the U.S. Bureau of Labor Statistics. GPI components and methods are summarized in Table 2. Detailed methods and data sources used to calculate each of the 26 GPI components are available as an online appendix at http://www.uvm.edu/giee/special/gpi.htm.

Table 2: Components and calculation methods for Northern Vermont GPI

GPI component	Contribution	Calculation method	Regional estimate
A. Personal consumption	+	Per capita income * national ratio of consumption expenditure to income	County level income data; national level ratio of consumption to income
B. Income distribution	+/-	(Gini coefficient in year/Gini coefficient in 1970)*100	County income distribution data
C. Consumption adjusted for inequality	+	Column A/Column B	Calculated
D. Household labor	+	Hours of housework based on gender and employment * hourly wage for domestic workers	Housework based on national figures using local employment and gender data; local domestic worker wage

			data
E. Volunteer work	+	Volunteer hours * average hourly wage rate	Volunteer hours based on national figure for volunteerism using on local education level data; local wage rate data
F. Household capital	+	Cost of consumer durables (item L) * depreciation rate of 12.5%	Consumer durables spending from item L; depreciation rate of 12.5% based on 8 year life span
G. Highways and streets	+	Stock value of highways and streets * 7.5% annual value	County level roads data; assuming 10% of net stock is the annual value and 25% of miles driven are commuting (defensive expenditure)
H. Crime	-	Direct costs of property crime + defensive expenditures to prevent crime	Local crime data * national level cost data; national level defensive expenditures scaled by population
I. Family breakdown	-	Cost of divorce + social cost of television viewing	County level divorce data * national cost data; local TV ownership * national viewing data * cost for families with children
J. Leisure time loss	-	Employment level * lost leisure hours * hourly average wage rate	County employment data; national leisure time loss data; local wage rate
K. Underemployment	-	# underemployed persons * unprovided hours/constrained worker*hourly average wage rate	Underemployment calculated using county unemployment data and national ratios; local wage rate
L. Cost of consumer durables	-	Per capita personal income * % spending on consumer durables	Personal income from item A; regional estimates of spending on consumer durables
M. Commuting costs	-	Cost of vehicles * % of vehicle use for commuting + cost of public transit + cost of	State level vehicle registration scaled by county population; county level transit expenditure

		commuting time	data; county level commute data*local wage rate
N. Household pollution abatement	-	Cost of automotive air filters and catalytic converters + cost of sewage and septic systems + cost of solid waste disposal	State level vehicle registration scaled by county population; county level housing served by septic and sewer; county level solid waste production*local and national cost estimates
O. Car crashes	-	# of crashes*cost per crash (property damage, health care, lost wages)	County level car crashes; national level cost estimates
P. Water pollution	-	County level water quality*benefit of unimpaired water	County level water quality data; national level cost estimates
Q. Air pollution	-	State level pollution data*population, forest, and farmland*cost/unit of air pollution damage to these assets	State level pollution data; county level population, forest, farmland data; national level pollution cost data
R. Noise pollution	-	Urbanization level*WHO estimate of noise pollution costs	County level urban population data; national cost data
S. Wetland loss	-	Total ha wetland lost*value/ha	County level wetland loss; global estimate of wetland value/ha
T. Farmland loss	-	Farmland ha lost to urbanization*estimated farmland value per ha	County level farmland loss; state and national level costs
U. Nonrenewable resource depletion	-	Total consumption of nonrenewable resources*cost to replace with renewables	State level energy consumption data
V. Long-term environmental damage	-	Tons of fossil fuel, wood, waste burned*marginal social cost of CO_2 emissions in a given	State level energy consumption data

		year	
W. Ozone depletion	-	Release of ozone depleting chemicals*cost/kg	Ozone depleting chemical production at national scale
X. Forest loss	-	Area of forest loss*forest ecosystem service value/ha	County forest cover data; global estimate for temperate forest value/ha
Y. Net capital investment	+/-	Scaled down national values based on population	Scaled down national values based on population
Z. Net foreign lending/borrowing	+/-	Not used; difficult to conceptualize at local scales	Not used; difficult to conceptualize at local scales

One notable change from Costanza et al. was our use of the methods of Talberth et al. (2007) to calculate long-term environmental damage costs. This method uses a \$89.57/ton CO_2 equivalent cost, based on a survey of recent studies (Tol 2005) on the economics of climate change. This value decreases in years prior to 2000, reflecting the increasing marginal costs of greenhouse gas emissions, and replaces the \$2.56/barrel "tax" on all forms of energy consumption. We believe this is a less arbitrary measure of the cost of climate change. Further detail on this component is provided in above-mentioned online appendix.

3.2 Results

We report our results similarly to Costanza et al. (2004), who grouped the 26 GPI components into eight functional groups: 1) Income (components A-C); 2) Households (components D, E, F, L, N); 3) Mobility (components G, M, O); 4) Social capital (components H-K); 5) Pollution (components P-R); 6) Land loss (components S, T, X); 7) Natural capital (components U-W); 8) Net investment (components Y, Z). Per capita results for these eight component groups are shown in Table 3.

Table 3: Summary indicators (per capita, U.S. 2000 dollars)

	Year	Income A,B,C	Households D,E,F,L,N	Mobility G,M,O	Social capital H,I,J,K	Pollution P,Q,R	Land loss S,T,X	Natural capital U,V,W
Caledonia County	1950	4,747	5,591	(253)	(1,223)	(1,414)	(2,446)	(2,252)
	1960	6,858	7,344	(460)	(917)	(1,134)	(2,521)	(2,492)
	1970	11,531	8,088	(291)	(614)	(1,062)	(2,457)	(4,442)
	1980	12,410	7,776	(392)	(1,278)	(390)	(2,174)	(4,306)
	1990	14,849	8,161	(290)	(1,627)	(653)	(1,975)	(4,389)
	2000	15,561	7,937	(471)	(2,283)	(465)	(1,776)	(5,297)
Chittenden County	1950	6,646	5,438	(1,109)	(1,207)	(1,782)	(259)	(2,252)
	1960	9,104	7,241	(1,573)	(865)	(1,331)	(267)	(2,492)
	1970	13,332	7,410	(1,646)	(600)	(1,096)	(234)	(4,442)
	1980	15,251	7,567	(1,772)	(1,316)	(370)	(230)	(4,306)
	1990	20,029	8,189	(1,544)	(1,728)	(497)	(258)	(4,389)
	2000	21,988	7,721	(1,609)	(2,495)	(421)	(251)	(5,297)
Essex County	1950	3,985	5,293	(122)	(1,165)	(1,931)	(1,879)	(2,252)
	1960	6,731	6,571	(288)	(895)	(1,581)	(1,721)	(2,492)
	1970	10,402	7,892	36	(605)	(1,682)	(1,754)	(4,442)
	1980	10,738	7,919	(78)	(1,168)	(390)	(1,526)	(4,306)
	1990	13,901	8,662	80	(1,557)	(653)	(1,484)	(4,389)
	2000	13,668	8,032	(369)	(2,196)	(465)	(1,420)	(5,297)
Franklin County	1950	5,365	5,434	(393)	(1,158)	(1,353)	(694)	(2,252)
	1960	8,056	7,061	(646)	(834)	(1,103)	(663)	(2,492)
	1970	11,833	7,537	(603)	(633)	(1,062)	(581)	(4,442)
	1980	13,991	7,315	(714)	(1,205)	(396)	(467)	(4,306)
	1990	17,804	7,846	(816)	(1,646)	(650)	(407)	(4,389)
	2000	19,207	7,400	(1,090)	(2,336)	(459)	(354)	(5,297)
Lamoille County	1950	6,186	5,293	(892)	(1,116)	(1,504)	(588)	(2,252)
	1960	9,783	7,579	(1,417)	(954)	(1,212)	(452)	(2,492)
	1970	12,006	7,679	(1,392)	(643)	(1,072)	(313)	(4,442)
	1980	15,093	7,651	(1,493)	(1,458)	(362)	(336)	(4,306)
	1990	18,549	8,377	(1,108)	(1,760)	(608)	(346)	(4,389)
	2000	19,132	7,748	(1,722)	(2,465)	(419)	(303)	(5,297)
Orleans County	1950	3,995	5,411	(153)	(1,111)	(1,521)	(2,394)	(2,252)
	1960	6,130	7,098	(316)	(837)	(1,269)	(2,354)	(2,492)
	1970	10,763	7,827	(93)	(601)	(1,210)	(2,223)	(4,442)
	1980	10,465	7,599	(181)	(1,211)	(425)	(1,993)	(4,306)

	1990	12,830	8,221	(141)	(1,648)	(725)	(1,956)	(4,389)
	2000	14,834	8,166	(224)	(2,222)	(501)	(1,759)	(5,297)
Washington County	1950	5,756	5,632	(694)	(1,232)	(1,255)	(459)	(2,252)
	1960	8,509	7,446	(1,072)	(876)	(987)	(420)	(2,492)
	1970	12.565	7,854	(1,107)	(573)	(929)	(363)	(4,442)
	1980	12,697	7,743	(1,214)	(1,269)	(346)	(362)	(4,306)
	1990	17,395	8,207	(972)	(1,671)	(573)	(371)	(4,389)
	2000	17,540	7,820	(1,041)	(2,438)	(421)	(365)	(5,297)
Vermont (state)	1950	7,289	5,458	(590)	(549)	(1,346)	(740)	(2,252)
	1960	8,804	6,805	(940)	(605)	(1,187)	(681)	(2,492)
	1970	12,586	7,715	(919)	(595)	(1,138)	(582)	(4,442)
	1980	14,124	8,210	(1,022)	(1,164)	(411)	(528)	(4,306)
	1990	17,029	7,971	(909)	(1,546)	(686)	(506)	(4,389)
	2000	18,338	7,763	(1,093)	(2,296)	(491)	(470)	(5,297)
United States	1950	7,734	4,739	(919)	(378)	(749)	(892)	(3,940)
	1960	9,360	5,969	(873)	(474)	(783)	(982)	(4,445)
	1970	12,989	7,282	(1,042)	(658)	(879)	(1,163)	(6,322)
	1980	15,680	8,231	(1,374)	(1,591)	(742)	(1,415)	(7,018)
	1990	18,474	8,272	(1,747)	(2,145)	(606)	(1,734)	(7,357)
	1997	19,088	7,765	(1,789)	(2,068)	(523)	(1,781)	(8,183)

Vermont's per capita GPI is greater than the U.S. average, with Chittenden County having the highest GPI of any Vermont county (Figure 2). Interestingly, GPI in the most rural counties (Caledonia, Essex, Orleans) was below the U.S. average in 1950 but had risen above the national average by 2000. For all Vermont counties, per capita GPI has increased at a faster rate than the U.S. average, suggesting that the growth in Vermont's consumption has not led to inequality, social, and environmental disamenities that have caused U.S. GPI growth to slow.

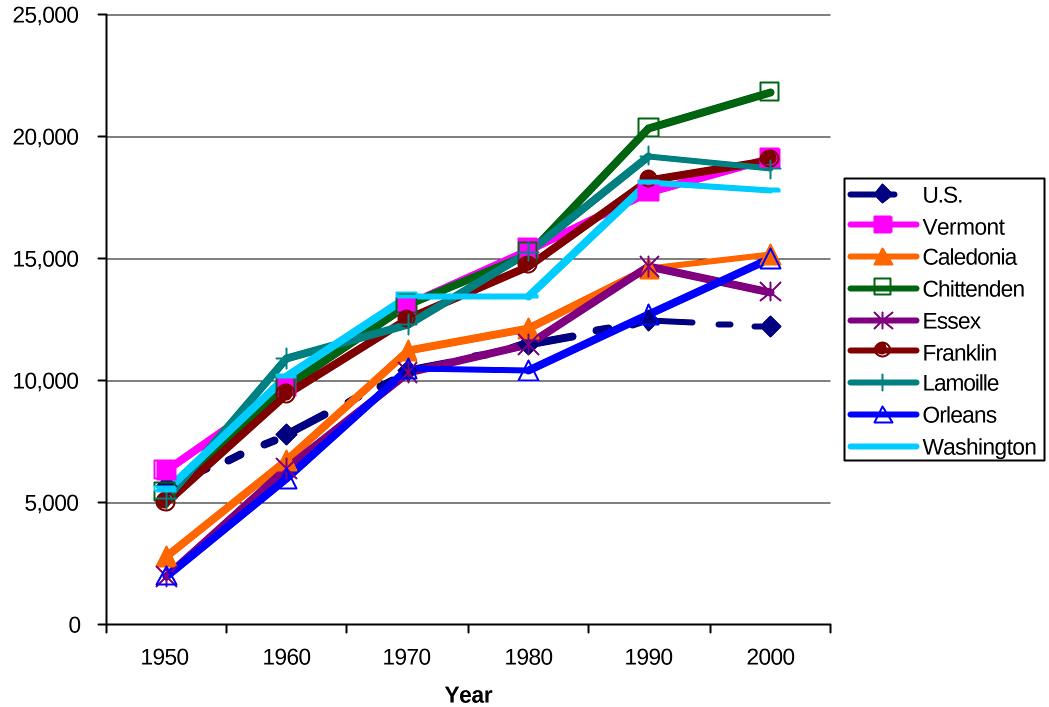

Figure 2: GPI per capita for U.S., Vermont, and seven Vermont counties for 1950-2000

Income in Vermont is generally below the U.S. average, with the exception of Chittenden County, but inequality is also below the U.S. average. This makes adjusted personal consumption expenditures in some counties approach the national level. Not surprisingly, the generally poorer rural counties have personal consumption levels below the state average.

County level household work and capital per capita did not differ greatly from national and state-level values. Costs and services of consumer durables are less in rural counties with lower incomes, as was the cost of household pollution abatement.

Mobility costs per capita are greatest in Chittenden County and least in the most rural counties. This reflects greater per capita value of services of highways and streets in rural counties combined with lower costs of commuting and crashes in highly rural areas.

Per capita social capital costs do not differ greatly among the six counties. Vermont has lower crime and family breakdown costs than the national average, but more underemployment and leisure time loss than the national average. Rural counties had consistently lower crime rates but higher costs of underemployment.

The per capita cost of pollution was actually greater than the national average in some Vermont counties. In some cases, this is due to high costs of air pollution damage to abundant forest resources, combined with low population. Thus, although rural Vermont counties may have low ambient pollution levels, per capita damage costs may be high.

Land loss per capita was similarly high for the most rural counties, owing to their small populations and high per capita costs of wetland and farmland loss. However, the rate of land loss has been slow since the 1950s, with per capita costs steadily decreasing.

Finally, per capita natural capital depletion was substantially less in Vermont than the U.S. average. This is driven by Vermont's below-average consumption of fossil fuels. Unfortunately, we lack county-level consumption data, so we cannot identify county-level consumption patterns in northern Vermont. We also note that using our method for valuing long-term environmental damage, the "threshold" effect that has been found at the national level in past U.S. GPI studies is less strongly seen. As the social cost of future emissions continues to rise, however, the threshold effect may become more evident. This highlights the importance of the cost of long-term environmental damage component in contributing to the overall GPI results.

3.3 Interpreting GPI results for northern Vermont

This study is the first local GPI calculation for a U.S. rural area. While we cannot generalize our findings to other larger urban-rural gradients or rural regions in the U.S., rural Vermont generally had lower income (hence, personal consumption), generated less solid waste, had less air, water, and noise pollution, and less forest and wetland loss. Due to their lower personal consumption, GPI was lower in rural counties than more urban Chittenden County, but rural counties lost less welfare due to environmental damage. This highlights the fact that GPI as an indicator remains largely driven by personal consumption. Other "highly influential" components that generally deducted or contributed $1,000 or more to per capita GPI included the value of household labor, services of household capital, leisure time loss, consumer durables spending, costs of air pollution, wetland loss, and depletion of non-renewable resources. "Minimally influential" components that deducted or contributed less than $100 to per capita GPI included volunteer work, cost of crime, water pollution, noise pollution, farmland loss, and ozone depletion (Figure 3).

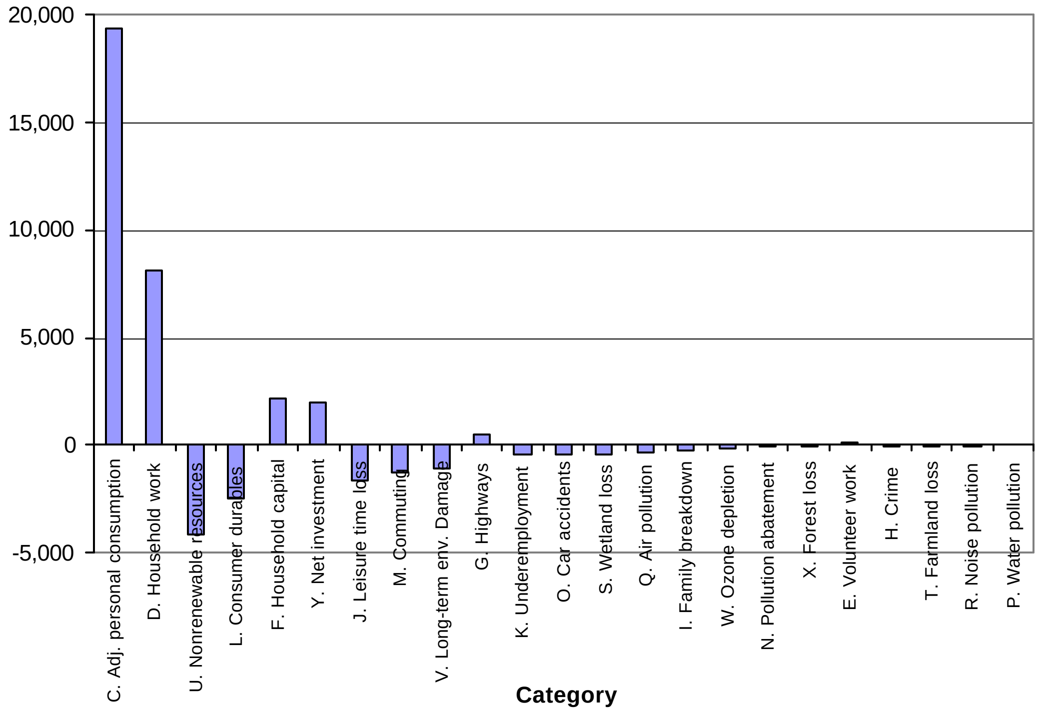

Figure 3: Relative contribution of components to Vermont county GPI

Unfortunately, data limitations obscure many of the local distinctions that we expect to exist in these rural counties. We believe social capital (including household labor, volunteer work, crime, and leisure time loss) may be greater in rural areas than urban areas. While rural areas have less crime, we did not have data for many other social components at the local level (hours of household labor, volunteer work, or on the job, and defensive spending to deter crime). For many components, surveys could be designed and implemented to obtain such data on rural versus urban quality of life. Lastly, rural areas showed substantially lower absolute levels of air, water, and noise pollution than urban areas. However, especially for air and noise pollution, the methods used were problematic and better applicable at the scale of larger urban areas.

Various non-monetary indicators have been developed to monitor socio-economic well-being in the Northern Forest. Non-monetary assessments provide an important benchmark for comparing and confirming general GPI trends. For example, the Northern Forest Wealth Index, an array of non-monetary indicators developed by the Northern Forest Center (2000), measures social capital as percent of registered voters. Voter participation was consistently higher in Northern Forest counties across the states of New York, Vermont, New Hampshire, and Maine during the 1998 general election. Similarly, lower income and lower property and violent crime rates were reported for all Northern Forest counties across the four states. The most comprehensive study comparing a Northern Forest region to more urbanized areas outside of the Northern Forest was conducted for the Adirondack region in New York state (Northup 1997). Not surprisingly, the study stressed the high quality of natural assets and social capital in the Adirondack Park, with the lowest crime and divorce rates in the state. Health care though, a crucial indicator that is not included in the GPI, scored lowest in this Northern Forest region than elsewhere in the state, with highest percentage of teen pregnancies and lowest number of physicians. Poverty levels were higher and the percentage of individuals holding college degrees was lowest. The above-mentioned studies faced difficulties in aggregating the values for all the indicators across multiple dimensions of well-being, a challenge to which GPI is well suited.

4. The value of GPI in estimating local well-being

4.1 GPI can highlight effects of local policy on well-being

Given the importance of regional and local policy on well-being, local GPI measurements can be used to identify areas where observed differences in components of well-being might result from certain series of policy choices. While it is difficult to ascribe changes in well-being itself or as measured by the GPI to any individual policy choice, GPI can identify components where a region is performing more strongly than nearby regions or the national average. In many regions, qualitative sets of well-being indicators are being developed. These indicators span the range of social, economic, and environmental performance or human, social, natural, and built capital. GPI can compliment existing indicators, offering an integrative, quantitative measure of well-being. One of the major strengths of using GPI is its ability to aggregate values using monetary figures as a common unit.

The GPI framework can be used to more comprehensively understand the consequences of local policy decisions (Matthews et al. 2003, Lawn and Clarke 2006). Generally, policies that effectively balance employment with protection of natural and social capital will produce positive GPI results. For example, the choice to develop local transportation based on new roads, highway construction, and automotive dependence might be viewed in a limited economic framework as providing beneficial temporary employment gains, along with increased mobility. Using the GPI framework, such policies might result in more car crashes, greater commuting time due to induced demand, increased air pollution, and loss of open space. Taking a more comprehensive view of such economic tradeoffs, the economic value of more compact development or reducing transportation demand might be shown to be a more desirable choice.

As another example, a county or state government seeking new revenue might consider increasing the payroll tax. Assuming that greater payroll tax reduces employment, this would be seen as a poor policy choice using a GPI framework. By comparison, ecological tax reform – taxing pollution, nonrenewable resource use, or land consumption while reducing the payroll tax (Repetto et al. 1992, European Environment Agency 2000), would likely be shown as a win-win situation for the economy and environment. The GPI would show this choice to improve local well-being. Although it is difficult to precisely predict changes in GPI resulting from such policy trade-offs, the GPI framework is useful for providing citizens, stakeholders, and policy makers with a more comprehensive understanding of the effects of such policy choices.

4.2 GPI can facilitate useful interregional comparisons

If applied carefully, interregional comparisons may be of value to researchers, policymakers, and citizens alike. Such comparisons can aid in assessing well-being as measured by the GPI, as well as in comparing certain GPI components. To make meaningful comparisons between two or more localities or regions, studies must use the same methods and have enough local data to avoid reliance on scaled down data that obscures local differences. Like GDP, total GPI will automatically be larger in economically larger regions as compared to small regions. Thus GPI should be compared on a per-capita basis.

The requirement of consistent methods presents problems when comparing local to national GPI. Local-national comparisons are useful to determine if a region is performing 'better' or 'worse' than the national average in terms of overall GPI or any of its constituent components. In the U.S., the Bureau of Economic Analysis tracks both national and state level GDP, showing a precedent for measurement and comparison at the state vs. national scale. As discussed in 5.2, certain GPI components are difficult to measure or conceptualize at the local scale. When making local-to-national comparisons, it may be best to drop the net capital investment and net foreign lending/borrowing components from both national and local estimates.

Differences in methods, components included, valuation techniques and data quality and availability mean that comparing local GPI between nations is not appropriate. Indeed, GPI researchers at the national level have rarely made comparisons between nations for these reasons, preferring to focus on trends in GPI and sometimes comparing changes in GPI to that of GDP.

It is also important to note that GPI does not explicitly account for interregional flows of nonmarket goods and services (Clark 2007). For example, Vermont lacks a heavy industrial base, and imports many manufactured goods from elsewhere. As such, the high forest cover and clean air and water Vermonters enjoy are partly achieved by importing more pollution-intensive goods from other parts of the world. These areas in turn experience greater environmental degradation associated with such industry. The presence of such interregional flows provides an interesting avenue for research in identifying the “winners” and “losers” associated with such flows in the global economy. Interregional trade in “virtual water” is one such example (Guan and Hubacek 2007), and regional GPI comparisons could aid in identifying such cases.

4.3 Government-collected data availability is improving

In the U.S., much of the data needed to calculate GPI are now compiled annually by state or federal agencies including the Census Bureau, Bureau of Economic Analysis, and Bureau of Labor Statistics. The Census Bureau also now compiles population data between decennial census years through the American Community Survey (ACS). Due to statistical constraints, annual ACS data will only be available for areas with population greater than 65,000. For smaller cities or counties, data will be aggregated across 2-5 year time scales, depending on the size of the community. While historical data may be lacking, this suggests that future local data quality and regularity may be greatly improved.

The American Time Use Survey, compiled annually by the BLS since 2003, improves time use availability, important for several components of GPI. Talberth et al. (2007) use ATUS data at the U.S. scale in their recent national GPI update. ATUS data can be broken down by state, and can be further broken down by county or zip code of the respondent. However, due to the relatively small sample size of the survey (13,000-21,000 respondents per year), it may be necessary to pool responses across years for areas with few respondents. ATUS is the first such continuous survey with a large enough sample size to be potentially useful at small geographic scales. For example, earlier time use surveys surveyed only 1,200-5,400 individuals and have some compatibility problems for time series analysis (Schor 1997). However new sources of time use data such as ATUS have the potential to greatly improve estimates of many costs and benefits important to the GPI at local scales.

Recent data for states, counties, and cities are often best obtained from state-level economic, transportation, labor, and natural resources agencies, many of which have improved their data collection and dissemination in recent years. Unfortunately this means that earlier annual data is often not available. Also, since data often come from different agencies, the most recent available dates may differ. For Vermont, the most recent local data for different GPI components dated from 2002 to 2006 as of completion of this study in late 2006. More responsive data collection and reporting by such agencies would improve the timeliness of calculated GPI estimates.

4.4 Incorporating commercially available data into GPI estimates

Time use, demographic, and environmental data used for the GPI are often calculated by government agencies, and are generally freely available. However, GPI also relies on consumer spending data for several important components. Such data are often compiled by national statistics agencies, but are rarely available at local scales. In the U.S. and likely in other industrialized nations, such data are compiled by commercial market research firms. These data are available for purchase. The cost of such data should be considered up front by researchers interested in estimating local scale GPI, particularly in the U.S. We note that commercial data are not included for our estimates of Vermont county-level GPI.

Such spending data are available for several critical items, including overall consumer spending (adjusted personal consumption), spending on energy (nonrenewable resource depletion and long-term environmental damage), consumer durables (and services of household capital), and indirect costs of crime (locks and security systems). Commercially available data greatly increases the number of variables with data available at the county level (Figure 4). Commercial data also represent four of the five most important components in terms of contribution to total GPI (Figure 3). When commercial data are combined with recent GPI estimates, the percent contribution to total GPI from local data sources rises from 11% to 75%, and scaled down national data falls from 67% to 24% of the total (Figure 5). Thus, most of the problems with data quality and availability concerns are successfully addressed, and high quality local GPI estimates are possible, using recent commercial data. We note that the highest quality GPI estimates will be from the year 2000 onward, which means that less confidence can be placed in the accuracy of time series data. Yet relatively high confidence can be placed in regional GPI comparisons for recent years that incorporate local consumer spending data. As an alternative to commercially generated data, it may also be possible to estimate overall consumer spending from local tax receipt data.

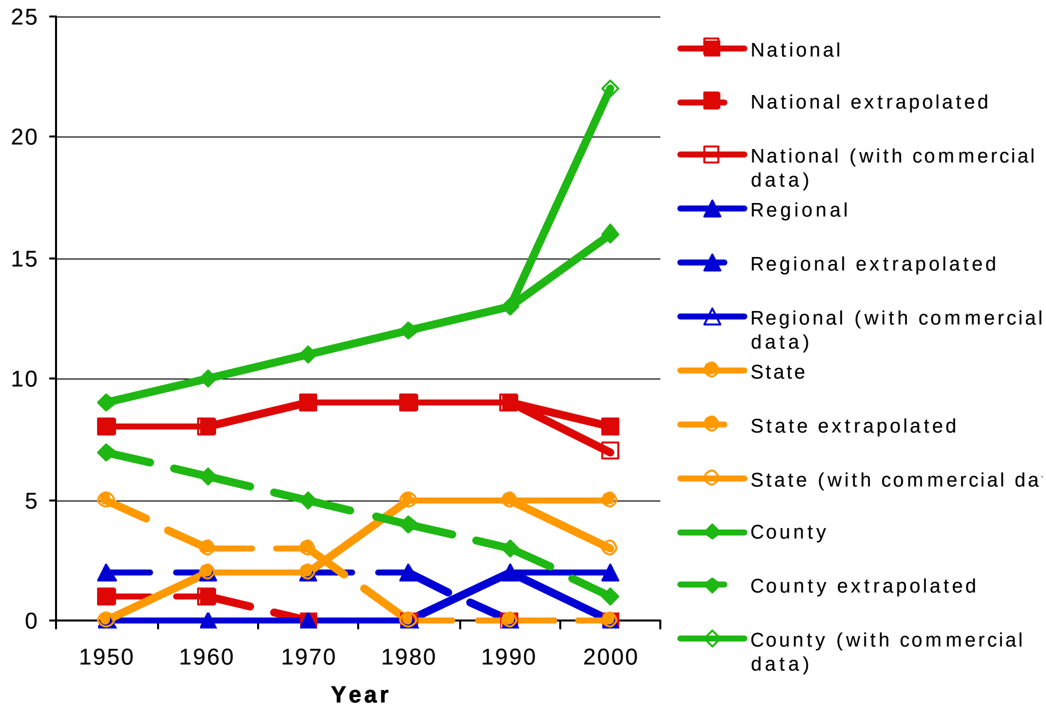

Figure 4: GPI data availability by spatial scale and year

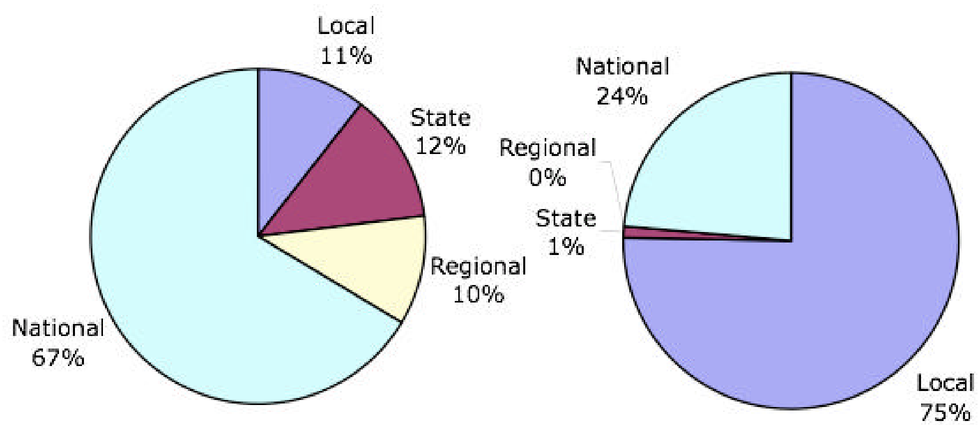

Figure 5: Percent monetary contribution to GPI by scale, without (left) and with (right) commercial data

5. Challenges in applying GPI at local to regional scales

5.1 Data limitations and confidence in historical data

Advocates and critics of the GPI alike acknowledge several components whose calculations rely on data or economic studies that are dated. In particular, calculations for the cost of air, water, and noise pollution date from the 1970s to early 1980s. More accurate, recent studies would considerably improve the estimates for these components. Local GPI studies can also incorporate valuation literature specific to the area that better reflects costs and benefits. As the nonmarket valuation literature grows more comprehensive, these local value estimates should improve. Lawn (2005) describes in more detail the need for consistent, consensus, and up to date valuation methods and overall GPI methods for the measure to gain wider acceptability.

As mentioned in section 2, socioeconomic and environmental data needed to calculate GPI are often lacking at local scales. Other authors of local GPI studies similarly report similar difficulty in obtaining local data (Matthews et al. 2003, Venetoulis and Cobb 2004, Jackson et al. 2006, Pulselli et al. 2006). In such cases, authors of past local studies have been forced to scale down national values based on population, land area, income, or other variables. These scaled estimates limit the value of local GPI studies, as presumed local patterns are lost. For our northern Vermont case study, data for several components were particularly poor at the local level, forcing us to also rely on scaled-down data. Time use data are generally poorly measured at the local scale, with the exception of commuting time. For components including hours spent at work, watching TV, performing household labor and volunteer work, we relied on national estimates.

For comparability with a previous study (Costanza et al. 2004), we estimated the GPI decennially for the period 1950-2000. The only other local U.S. GPI study, Venetoulis and Cobb (2004), estimated local GPI for the San Francisco Bay area in 2000 only. As an integrated quality of life indicator, it is desirable to track GPI on a finer temporal time scale than every ten years. Yet data from earlier decades are generally of lower quality. For many components, data from earlier decades are available only at coarser spatial scales or simply do not exist (Figure 4). In many cases, we extrapolated data backward to obtain values for earlier decades. This gives less confidence in GPI estimates from earlier decades, especially prior to 1980. However, from 1990 onward we obtained actual data for more components of interest. This is likely due in part to the growth of the Internet in the 1990s, and the fact that online data are relatively abundant from the 1990s onward and relatively scarce prior to 1990.

Because of these data limitations, we place relatively greater confidence in local GPI estimates from 1990 onward. Unfortunately, even our recent estimates relied heavily on scaled down national and state level data (Figure 4). This was especially true for the components that contributed the most value to GPI, including personal consumption, household work, and net investment (scaled down from national values), nonrenewable resource depletion and long-term environmental damage (scaled down from state values), and consumer durables and household capital (scaled down from regional values). Because of this scaling, even our year 2000 values must be interpreted with caution. However, the growing availability of commercially available market research data has the potential to provide accurate, local values for many of these important components, as discussed in section 4.4.

5.2 Relevance at local scales

Conceptually, some GPI components are difficult to reconcile at the local level. The most problematic components are net capital investment and net foreign lending/borrowing. Reporting on net capital investment is often incomplete at local scales for two reasons. First, statistics for local companies with few competitors are not reported, in order to protect these companies' privacy. Second, data are not reported for companies with fewer than 950 employees. These restrictions on investment data would be less problematic in areas with larger populations and industrial activity. Net foreign lending and borrowing data are unavailable at local scales, and raises questions with the definition of "foreign" versus "local" economic activity. On one hand, the local share of national debt could be used for this component; on the other hand investment inside or outside a state could be included as a measure of economic self-sufficiency. In either case, the appropriateness of measuring net foreign lending/borrowing at local scales is questionable. Bleys (2008) argues that inclusion of net capital investment and net foreign lending/borrowing are inappropriate for inclusion in ISEW/GPI, and that these components are better treated as supplemental accounts.

Transboundary impacts must be carefully considered when comparing GPI results at any scale, but are particularly important at regional scales for various types of pollution and natural resource depletion. Pollution or natural capital loss in one region can substantially affect well-being of adjacent regions. In Vermont, the acid deposition produced from emissions of coal-fired power plants in the Midwestern U.S. is a good example. Here the costs are borne by users of Vermont's lakes and forests, while the benefits accrue to Midwestern consumers who receive cheaper electricity. To better measure economic well-being, 'genuine progress' studies for the Midwestern U.S. should bear responsibility for these costs (Clarke 2007). Since pollution and natural capital depletion ignore artificial boundaries, regional GPI studies should take care in accounting for the impacts of pollution and depletion.

This loss of forests, wetlands, and farmland may lose relevance at small spatial scales, especially in rural areas where natural capital is not scarce. Similarly, noise pollution might be a relatively irrelevant factor in remote, forested areas. Land loss and pollution often capture the economic costs of poorly-planned development. Yet in rural settings such as northern Vermont, where development is minimal and small towns are surrounded by forests, farmland, and wetlands, the social costs of such loss are likely to be much less than in rapidly urbanizing areas experiencing massive loss of open space.

Finally, when GPI is presented on a per capita basis, sparsely populated areas may have unexpectedly high costs for certain components. For example, Essex County, Vermont had the greatest value for services of streets and highways as well as the greatest cost of air pollution, because these attributes were similar for all counties but population itself was much smaller. As such, population differences may produce unexpected per capita results.

6. Conclusions

The GPI and ISEW, originally developed for use at the national scale, are increasingly being estimated at local and regional scales. Such studies are an outgrowth of growing interest in measuring and promoting quality of life or well-being through sets of indicators as well as integrative measures like GPI. Based on a case study in northern Vermont, we conclude that there are important limitations in data quality and methods that should be strongly considered before applying GPI locally. In particular, when local data is lacking, national or state level estimates must be used and 'scaled down,' reducing the comparative value of local estimates. In the U.S., this problem is most pronounced prior to the 1990s. Data availability and quality has improved in recent years and with widespread use of the Internet for disseminating local socioeconomic and environmental data. By incorporating commercial data on consumer spending patterns, researchers can compile most of the needed data at local scales, leading to robust GPI estimates, particularly for recent years. Such estimates can provide useful means of tracking local well-being, and can also facilitate interregional comparisons. We expect that data quality and availability are likely to be better in industrial nations than developing nations. As such, researchers interested in developing local GPI estimates should carefully evaluate that region's data sources to ensure that GPI figures best reflect the unique local conditions that contribute to societal well-being.

Acknowledgements

Support for this project was provided by the Northeastern States Research Cooperative grant "Monitoring socioeconomic well-being in the Northern Forest". We are grateful to numerous individuals, primarily at the University of Vermont and within various agencies of Vermont's state government for providing assistance with data acquisition and discussion of methods. Brent Bleys, Phil Lawn, John Talberth, and two anonymous reviewers provided valuable suggestions to improve this paper.

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CHAPTER 4: THE GENUINE PROGRESS INDICATOR AS A MEASURE OF LOCAL AND REGIONAL ECONOMIC WELFARE: A CASE STUDY FOR NORTHEAST OHIO[1]

Abstract

Although the Genuine Progress Indicator (GPI) and related Index of Sustainable Economic Welfare (ISEW) have been estimated at the subnational level, these estimates often rely on poor quality data and have rarely enabled intra- or inter-regional comparisons. We calculated the GPI for the State of Ohio, cities of Akron and Cleveland, and 17 Northeast Ohio counties for the years 1950-2005. These estimates use the highest quality data yet for a U.S. local study, and particularly for 1990-2005 can be considered robust estimates. We evaluated temporal and spatial GPI trends, including inter- (Ohio versus Vermont) and intra-regional (urban-suburban-rural) comparisons. From 1990-2005, we found that per capita GPI grew in eight counties but declined for nine counties, Ohio, Akron and Cleveland. Per capita GPI was greatest in suburban counties and lowest in urban areas, and was greater in Vermont than Ohio. These trends are largely driven by gains in personal consumption relative to rising environmental, social, and economic costs. Important costs include those of income inequality, climate change, nonrenewable resource depletion, and consumer durables. Sensitivity analysis shows that the local datasets used in this study played an important role in producing reliable results. Results are also greatly influenced by the

[1] Bagstad, K.J. and M.R. Shammin. In preparation. The Genuine Progress Indicator as a measure of regional economic welfare: A case study for Northeast Ohio. Target journal: Ecological Economics.

assumptions that go into the calculations. Because GPI is so sensitive to changes in personal consumption versus other costs, it functions neither as a measure of strong sustainability nor as a perfect measure of social welfare. Yet consistently calculated local GPI estimates for different parts of a country can show how the costs and benefits of economic growth are distributed within a country, while engaging the public and decision makers in discussions about economic, social, and environmental goals and policies. Local academic and nonprofit organizations are using the GPI framework to advance discussions about sustainability and economic development in Northeast Ohio.

Keywords

Genuine Progress Indicator, Index of Sustainable Economic Welfare, urban-rural, social welfare, quality of life, regional development

1. Introduction

1.1 The GPI framework

As communities of all sizes have become more interested in measuring and promoting quality of life, the use of indicators for community well-being has grown (Haggerty et al. 2001). Numerous communities have developed suites of quality of life indicators (Sustainable Measures 2006) yet face the problem of creating an index from indicators with dissimilar units, such as rates of farmland loss, voter registration, and crime. At the national, state, and metropolitan area scales, gross domestic product (GDP) is still frequently used as a proxy measure for society's welfare, despite its many flaws (van den Bergh 2009). These have become evident through a growing body of literature suggesting that increased wealth and economic output alone do not always improve quality of life or subjective well-being for individuals or society (Diener et al. 1999, Eckersley 2000, Frey and Stutzer 2002, Kahneman 2004, Easterlin 2005). Economists and politicians, including some of the original architects of GDP accounting have also noted the misuse of GDP as a welfare measure.

Since at least the late 1960s, economists have attempted to adjust GDP to better measure society's well-being. In perhaps the most well-known of these early studies, Nordhaus and Tobin (1972) concluded that as of the early 1970s, economic growth was leading to improvements in quality of life in the United States. Daly and Cobb (1989) revisited Nordhaus and Tobin's findings with their Index of Sustainable Economic Welfare (ISEW), which was later revised as the Genuine Progress Indicator (GPI). The ISEW/GPI (hereafter referred to as GPI) begins with a measure of personal consumption, weighted to account for income inequality, and deducts or adds value for various monetized measures of built, human, social, and natural capital. This can be expressed in the form of the equation (adapted from Hanley et al. 1999):

$$GPI = C_{adj} + G + W - D - S - E - N \quad (1)$$

Where: C_{adj} = personal consumption adjusted to account for income distribution, G = growth in capital and net change in international position, W = non-monetary contributions to welfare (e.g., household labor, volunteer work), D = defensive private expenditures, S = depletion of social capital (e.g., cost of crime, family breakdown, lost leisure time), E = costs of environmental degradation, and N = depletion of natural capital.

The inclusion of these components makes GPI better suited than GDP to addressing questions of distribution, societal well-being, and sustainability within the economy. Daly and Cobb and subsequent authors found that GPI grew, though not as quickly as GDP, until the mid-1970s, and has since leveled off or declined slightly. These results agreed with Max-Neef's (1995) "threshold hypothesis," which states that economic growth improves quality of life up to a point, but eventually erodes environmental and social quality, reducing quality of life. Studies in numerous other nations corroborated these findings (Jackson and Stymne 1996).

Several components of GPI's theoretical framework have been questioned (Neumayer 1999, Neumayer 2000). Part of this controversy relates to what GPI is intended to measure – is it an indicator of sustainable (Hicksian) income, a pure replacement for GDP, an index of economic welfare, or an assessment of how well human needs are met? Lawn (2003) provides a theoretical basis for using GPI as an improved measure of welfare over GDP. Lawn uses a definition of income derived from Fisher - the utility or satisfaction consumers get from the economy - as opposed to Hicks' definition of income as the maximum a household or nation can consume without reducing its ability to do so in the future. However, Harris (2007) notes that these alternative views of income are typically misunderstood, and that both Hicksian and Fisherian income are consumption-based. Since Hicksian income is concerned with both present and future income, while Fisherian income focuses only on present income, Harris argues that Hicksian income is a better lens for evaluating sustainability (although many ecological economists might prefer Fisher's concept of income as "psychic flow" or utility, versus Hicks' concept of income as monetary-based). Further, Harris argues that there is a lack of evidence that the disamenities measured by the GPI are always caused by economic growth, or that transition to a steady-state economy is the only way to increase the GPI.

Neumayer (1999) also notes that because GPI aggregates the value of built, human, social, and natural capital, it does not serve as an indicator of strong sustainability. Since natural capital could be liquidated to increase consumption with potentially increasing GPI, for instance, GPI does not serve as a measure of strong sustainability. Despite these problems, Ziegler (2007) argues that the GPI has its greatest value as a "debunking index" useful in showing the limitations of the still-entrenched mindset of measuring and promoting GDP growth.

Like nations, sub-national political jurisdictions of all sizes are increasingly interested in measuring quality of life, and in developing policies to support social well-being. At least in industrialized nations, these economies are open and often small. Many fiscal, social, and environmental policy choices take place at the national level, and greatly affect well-being at local scales. Yet local jurisdictions also make similarly important decisions. For example, a state or province could use tax policy to encourage or discourage employment in certain economic sectors. A county or municipality can make land use or resource extraction decisions that liquidate natural capital in favor of sometimes short-term employment gains. In other cases, local jurisdictions may choose to adopt more stringent environmental or social goals than the national mandate. Since GPI aggregates a broad suite of economic, social, and environmental indicators, it can be used to compare well-being in between two or more regions with different policies. GPI's ability to aggregate an otherwise diverse set of indicators is an important strength as a measure of well-being.

1.2 Local measurements of GPI

Although GPI was originally developed as a national-level macroeconomic indicator, the costs and benefits of economic growth are not distributed evenly across a nation. Understanding these local and regional differences has been a justification for developing local-scale GPI studies. In the U.S., GPI has been estimated locally for Minnesota (Minnesota Planning Environmental Quality Board 2000), the San Francisco Bay area (Venetoulis and Cobb 2004), and Vermont (Costanza et al. 2004, Bagstad and Ceroni 2007). These studies found GPI to be consistently higher than the national average for these areas. This may be due to efforts in these regions to develop strong local economies while preserving environmental quality and social cohesion.

Outside the U.S., local GPI and ISEW studies have been conducted in Australia (Lawn and Clarke 2006), Canada (Anielski 2001), China (Wen et al. 2007), Italy (Pulselli et al. 2006, Pulselli et al. 2008), and the U.K. (Moffatt and Wilson 1994, Matthews et al. 2003, Jackson et al. 2006). These studies do not always provide comparisons to national-level figures, often due to differing data or methods. They also reveal limitations with using GPI at local or regional scales. For industrialized nations in particular, most data needed for the GPI are readily available at the national level. This is often not the case for local jurisdictions. In the U.S., historical data may be available only for decennial census years, or may simply not exist. In such cases, analysts have scaled down national or state values based on variables such as population or land area. While this method provides "filler" estimates, it also obscures the local differences that influence well-being, the main justification for undertaking these studies. Bagstad and Ceroni (2007) describe how to maximize the use of local data in GPI studies, particularly in the U.S.

Since GPI was developed as a national-scale indicator, local GPI studies face other limitations (Clarke and Lawn 2008). These include data availability and the need for consistent data sources and methods, the fact that GPI does not account for cross-boundary impacts of manufacturing, energy production, or resource extraction (Clarke 2007), and the fact that local governments do not have full power to set policy related to all of the GPI's component indicators. The first limitation can be overcome with careful and consistent data collection and management. The second limitation should be recognized, but can be addressed regionally by examining trends in GPI across urban to rural environments. As for Clarke and Lawn's third limitation, state and local governments in the U.S. do have important policymaking powers in regards to land use planning, energy use, and other relevant GPI components. This illustrates the potential value in using the GPI as a local and regional decision support tool, provided that estimates are accurate, timely, and reflect changing local conditions from year to year.

Finally, given GPI's value as a "debunking index" that exposes the limitations of GDP (Ziegler 2007), its use at local scales is just as relevant as at national scales. Given the lack of dialogue in the U.S. about alternatives to GDP since the mid-1990s (Cobb et al. 1995), this discussion may be more fruitful at the local level than the national level.

1.3 Objectives

In this paper, we expand on the work of Bagstad and Ceroni (2007) in developing the GPI for local and regional applications, particularly in the United States. Our objectives were to: 1) enable spatial and temporal GPI comparisons across a large urban-rural gradient and between regions within the U.S., 2) use sensitivity analysis to explore how use of local data and improvements to methods impact local GPI estimates, and 3) to develop and describe a nascent policy process for incorporating the GPI in local decision making. Sections 2, 3, and 4 provide the methods, results, and discussion, respectively for a GPI case study of the State of Ohio, cities of Akron and Cleveland, and 17 counties in Northeast Ohio. Section 5 describes a local policy process beginning in Northeast Ohio that uses the GPI to enable a more comprehensive understanding of regional well-being.

2. Methods

2.1 Study area

Our study area included a seventeen-county region in Northeast Ohio (Figure 1), stretching along Lake Erie from the Pennsylvania border to Sandusky Bay. This encompasses about 20,700 km^2, or 19.5% of Ohio's land area. The region's largest city is Cleveland, an important center for manufacturing and Great Lakes shipping. The area also includes a number of other medium-sized industrial and port cities, including Akron, Canton, Lorain, and Youngstown. Like many Midwestern manufacturing centers, these cities have lost population in recent decades. For example, Cleveland's population fell from 914,808 in 1950 to 449,995 in 2005. The causes of population loss include declines in manufacturing employment due to movement of industries from the U.S. to nations with cheaper labor costs, migration with the region from cities to the suburbs, and migration out of the region from the northern "rust belt" to the southern "sun belt." Regional population has remained relatively stable from 1960 to the present, however, ranging from 4.1 to 4.5 million. Against a backdrop of population migration and manufacturing decline, there has been growing interest in improving regional sustainability. Efforts underway include retooling of local industries to service the growing renewable energy industry and strengthening local agriculture and food systems.

Figure 1: Study area map

Outside of Northeast Ohio's urban areas, land use was historically dominated by agriculture. Recent decades have seen both suburban expansion on the urban fringe and regrowth of forest cover following abandonment of marginal agricultural land. Statewide, agricultural land declined from 80% to 57% of all land area from 1950-2005 while forested land expanded from 19.5% to 30%. Northeast Ohio has seen similar trends in agricultural and forest cover change. Counties near Cleveland, such as Geauga, Lake, Lorain, and Medina have seen substantial population growth in recent decades, primarily as agricultural land and forests are converted into developed land. Outside these suburbanizing counties, however, land use today is still predominantly agricultural.

2.2 Construction of GPI estimates

To calculate the GPI for Northeast Ohio, we strove to maintain consistency with past local studies, following the methods of Bagstad and Ceroni (2007), who in turn follow Costanza et al. (2004). Methods for the services of household capital, cost of consumer durables, climate change, and ozone depletion follow Talberth et al. (2007), the most recent U.S. GPI study (Table 1). When improved data sources or methods were available, we noted these changes in an online Appendix at http://www.uvm.edu/giee/?Page=genuine/index.html. This appendix describes detailed data sources, methods, and assumptions. We briefly describe major changes to our methods below. Following past studies, we calculated values for the decennial years 1950-2000, but also for the year 2005, for all 26 GPI components. We converted monetary values into year 2000 U.S. dollars using the Consumer Price Index (CPI) from the U.S. Bureau of Labor Statistics. For the years 1990-2005, we used local CPI data for the Cleveland-Akron urban area as a deflator. We used national CPI values from 1950-1980, where local data were unavailable.

Table 1: Methods, local data, and assumptions used for GPI components

GPI component	Methods followed	Additional local data used	Assumptions for conservative estimates
A Personal consumption	B & C (2007)	Local consumer spending data	
B Income inequality	B & C (2007)		
D Household labor	B & C (2007)	Local time use and wage rate data	
E Volunteer work	B & C (2007) plus state-level estimates hours worked, value of volunteer hours		
F & L Services of household capital,	T et al. (2007)	Local consumer spending data on	

costs of consumer durables		consumer durables	
G Streets & highways	B & C (2007)		
H Crime	C et al. (2004)	Local consumer spending data on security systems	Costs of murder not estimated (vs. estimated using costs used for traffic fatalities)
I Family breakdown	B & C (2007)	State time use data on TV watching	
J Leisure time loss	B & C (2007)	Local wage rates and time use for work hours	
K Unemployment & underemployment	C et al. (2004)	Local wage rates	
M Commuting	C et al. (2004)	Local wage rate for commute time	
N Household pollution abatement	C et al. (2004)		
O Vehicle crashes	B & C (2007)		
P Water pollution	B & C (2007)		
Q Air pollution	B & C (2007)		
R Noise pollution	C et al. (2004)		
S Wetland loss	B & C (2007)		Only wetland losses since 1950 valued
T Farmland loss	B & C (2007)		
U Nonrenewable resource depletion	B & C (2007), M (2007)		Renewable energy sources correctly substituted: wind and solar for electricity, biofuels for liquid fuels
V Climate change	T et al. (2007), S & B (in review)	Local consumer spending data on energy	Local data on the size of manufacturing, commercial sectors, household expenditures on energy
W Ozone depletion	T et al. (2007)		Costs not accumulated
X Forest loss	C et al. (2004), forest valued at $481/ac-yr		Only forest losses since 1950 valued
Y Net capital investment	C et al. (2004)		
Z Net foreign lending/borrowing	C et al. (2004)		

C et al. (2004): Costanza et al. (2004)
B & C (2007): Bagstad and Ceroni (2007)
T et al. (2007): Talberth et al. (2007)
M (2007): Makhijani (2007)
S & B (in review): Shammin and Bullard (in review)

For the cost of crime, which previously only included the cost of property crime, we tested the impact of adding the cost of lost human life due to murder and manslaughter. Since the GPI already uses a conservative value of human life estimate for vehicle crashes, we applied the same value to the loss of life due to violent crime.

For wetland and forest loss, we compared the value of forest and wetland loss since pre-settlement times versus loss since 1940 only. The use of pre-settlement forest and wetland cover estimates is problematic for two reasons – the high uncertainty associated with the estimates, and the limited policy relevance for social well-being when using a pre-settlement baseline for natural areas cover.

For the cost of nonrenewable resource depletion, we used replacement costs for energy consumption at local scales. This cost accounts for the need for communities to transition to renewable energy, a topic of serious discussion for a growing number of communities. We divided energy consumption into transportation fuels that could be replaced using biofuels versus other consumption including electricity generation, which could be replaced by sources such as wind and solar. While there is much uncertainty surrounding technologies to scale up renewable energy use and their costs, Makhijani (2007) provides cost estimates for the widespread adoption of renewable energy sources.

For the cost of climate change, we compared two methods: 1) using methods from Talberth et al. (2007), but scaling down state-level CO_2 emissions data by sector using manufacturing employment, commercial employment, household spending on energy, and vehicle miles traveled for the industrial, commercial, household, and transportation sectors respectively; and 2) using Shammin and Bullard (in review), which assigns greenhouse gas emissions intensities to different categories of consumer spending, and multiplying these intensities by the spending totals for each spending category. Intensities are based on U.S. Department of Commerce Economic Input-Output Life Cycle Analysis (EIO-LCA) database (EIO-LCA 2006). These intensities include carbon emissions associated with both direct energy consumption (natural, gas, electricity, gasoline, etc.) and also indirect energy consumption (energy embodied in various goods and services, Shammin et al. in press). Table 2 summarizes the intensities used for our GPI calculations. By accounting for the CO_2 emissions intensities of consumption, we avoid the "open economy" problem in the GPI, where costs of consumption decisions are not borne locally (Clarke 2007). We did not accumulate the cost of CO_2 emissions, a controversial practice when estimating the GPI (Neumayer 2000).

Table 2: Energy and carbon intensities for aggregated personal consumption categories for Northeast Ohio (2005)

Personal consumption categories	**Energy intensity (btu/$)***	**Carbon intensity (lbs/$)****
Food at Home	6,155	0.25
Food Away From Home	4,753	0.20
Alcoholic Beverages	4,556	0.18
Household Operations: Personal services	2,522	0.10
Household operations: All other	3,998	0.15
Owned Dwelling – Mortgage interest	1,852	0.08
Owned Dwelling – Mortgage principle	7,203	0.32
Owned Dwelling - Property tax	0	0.00
Rented dwelling	3,450	0.14
Other lodging	4,651	0.19
Natural Gas	114,710	2.80
Electricity	151,750	4.70
Fuel Oil	111,300	3.66

Bottled Gas	111,300	3.66
Coal/Wood/Other	111,300	3.66
Phone	2,356	0.09
Water/Sewer	8,031	0.32
Housekeeping Supplies	4,589	0.18
Household Furnishings & Equipment	4,688	0.19
Apparel & Services	5,926	0.24
New cars, trucks, vans	5,984	0.24
Used cars, trucks, vans	6,470	0.26
Other vehicles	8,769	0.34
Gasoline	94,299	3.20
Diesel	94,299	3.20
Motor Oil	94,299	3.20
Other vehicle expenses	2,219	0.09
Public trans	18,128	0.71
Air	18,128	0.71
Health care	1,799	0.73
Entertainment-Reading	3,554	0.14
Personal care	3,500	0.14
Education	2,689	0.11
Tobacco	1,604	0.07
Cash Cont	3,346	0.14
Life/other insurance	1,424	0.06
Miscellaneous expenses	3,809	0.15

* Authors' calculations based on Shammin et al. (in press)
** Authors' calculations based on Shammin and Bullard (in review)

Finally, we compared the cost of ozone depletion using accumulation (as opposed to nonaccumulation) of costs from year to year. We did this for ozone depletion but not CO_2 emissions because emissions of ozone depleting chemicals have basically stopped, while their social costs have not. CO_2 emissions and their social costs, however, both continue to rise.

Local data quality relevant to the GPI has improved considerably in recent years (Bagstad and Ceroni 2007). The American Community Survey now provides annual socioeconomic data for cities and counties with populations of 65,000 or more, along with pooled data over multiyear periods to provide estimates for cities and counties with smaller populations. ACS estimates contain a larger margin of error than the decennial census. This margin of error is due to ACS' smaller sample size, as it surveys only 1 in 40 households, versus 1 in 6 during the 2000 Census.

The Bureau of Labor Statistics' (BLS) American Time Use Survey (ATUS) can provide state-level time use data, enabling local adjustments to national time use data. Local wage rates from the BLS are also used in this study for all relevant GPI components. Finally, we included detailed county-level consumer expenditures data purchased from ESRI (ESRI 2008). These data are available from the BLS at national and regional scales, but are unavailable in the public domain at the state, county, or city level. As such we purchased consumer spending data on over 750 goods and services and combinations of goods and services for use in our analysis. Cumulatively, the use of these local data gives us confidence that results reflect local conditions for GPI components more so than past studies, which relied on less accurate, often scaled down data.

2.3 Temporal comparisons

We estimated the GPI for decadal years from 1950-2000, along with the year 2005. This enables us to compare time trends in the GPI. However, as noted by Bagstad and Ceroni (2007), considerably more confidence can be placed in GPI estimates from recent years, particularly from 1990 onward. This is because data from earlier years are often scaled down from state or national values or is extrapolated backward from more recent trends. As such we focused our analysis primarily on the 1990-2005 time series.

2.4 Spatial comparisons

Spatial comparisons can take place at an intra- or inter-regional scale. To evaluate results within the region, we can compare individual counties to each other or group counties as urban, suburban, and rural (Figure 2). Although these distinctions are subjective, they address the changing ecological and socioeconomic setting between a region's urban centers and rural hinterlands. We classified counties as urban, suburban, or rural based on 2005 population density and proximity to urban centers. Cleveland, Akron, and Cuyahoga County have Northeast Ohio's highest population densities, and we thus classified these as urban. Summit County (containing Akron) is substantially less dense than Cuyahoga, and is primarily suburban. Five counties surrounding Cuyahoga and Summit (Geauga, Lake, Lorain, Medina, Portage) have greater population density and urban proximity than the rural counties, and we designated them as suburban. Finally, the smaller cities of Canton and Youngstown support their own suburbs, along with county populations too large to be truly rural. As such we included Mahoning, Stark, and Trumbull counties as suburban. The remaining seven counties meet neither of these criteria for urban proximity or population density, and we designated these as rural. Some of the counties classified as "suburban" have substantial rural areas contained within them, while some of the "rural" counties contain one or more dense but small cities, though these cities generally lack suburbs. Also, some of the "suburban" counties had much more rural character prior to the last half century of suburbanization. As such it would be inaccurate to have called some of them suburban in earlier decades.

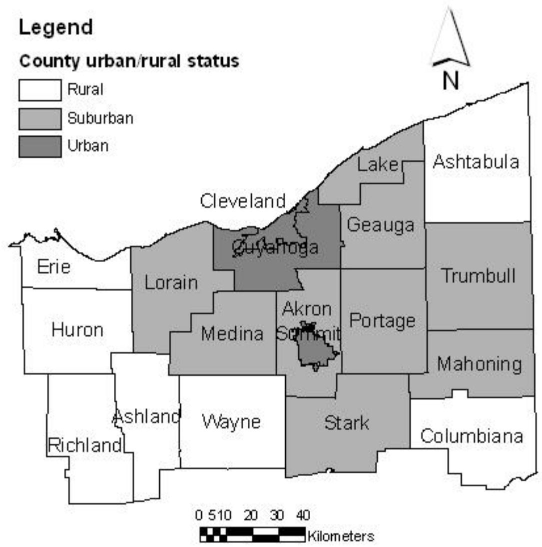


Figure 2: Urban, suburban, and rural counties

For inter-regional comparisons, we first adjusted the value of Bagstad and Ceroni's (2007) Vermont GPI estimates to ensure comparable methods and data sources. This allowed us to compare GPI results between Vermont and Ohio. We compared GPI at the state level (Ohio versus Vermont), for rural counties (average of seven rural Northeast Ohio counties versus six rural Northern Vermont counties), and for Chittenden County, Vermont's most urban county, versus an average of nine suburban Northeast Ohio counties. Since Vermont has no urban centers on the scale of those found in Northeast Ohio, urban-urban comparisons were not possible. As local

GPI studies are completed elsewhere in the U.S., further inter-regional comparisons will be possible, assuming that researchers use consistent methods and data sources. Regrettably, inconsistencies in data and methods preclude comparing results from Ohio and Vermont to earlier studies in Minnesota and the San Francisco Bay Area.

2.5 Sensitivity analysis

We used different data and assumptions to estimate GPI with above-described local data versus non-local data, which is often scaled down from national or state values (Table 1). Since added time and expense are needed to incorporate local data, we tested the relative value that local data adds in changing the overall estimates. Alternative data and assumptions described in section 2.2 were also examined for the valuation of human life and natural capital, which value these resources using more or less conservative methods.

3. Results

3.1 Temporal trends

Given the complexity of reporting all GPI components for 20 unique geographic areas across multiple decades, we report detailed results as supplemental online material at http://www.uvm.edu/giee/?Page=genuine/index.html. We report per capita GPI values here using figures or tables as appropriate. Similar to national GPI trends (Talberth et al. 2007), GPI in most of Northeast Ohio rose for a period, in this case 1950-1990. Since 1990, GPI has remained stable, declined, or increased in different geographic areas (Figure 3). Regrettably, GPI values for earlier decades have large associated uncertainty due to the lack of local data. When local data is unavailable, extrapolation is required in order to obtain estimates for these early decades (Bagstad and Ceroni 2007). As such, we resist the temptation to present and evaluate GPI trends from 1950-1980, the period where data are generally much less reliable. We instead focus our analysis of temporal trends for the years 1990-2005.

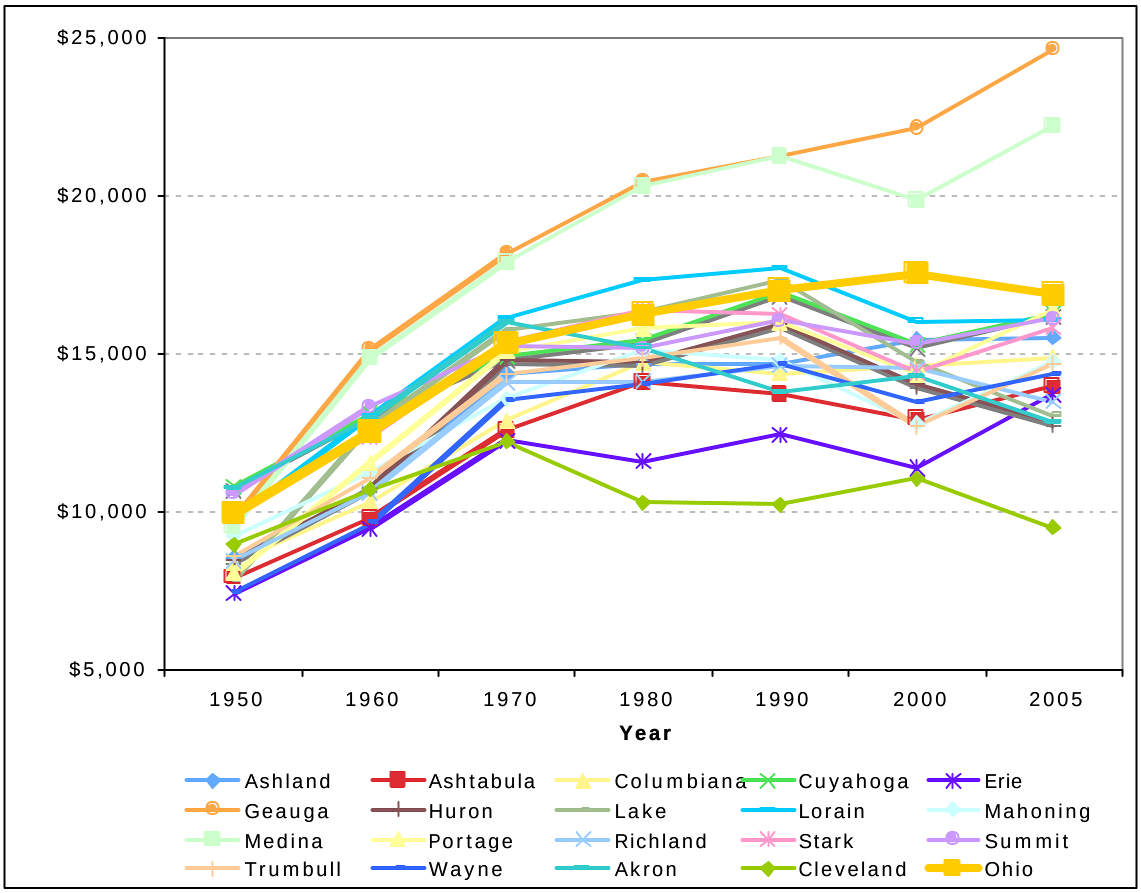


Figure 3: Northeast Ohio GPI trends from 1950-2005

From 1990-2005, per capita GPI grew in eight counties and declined in nine counties, the cities of Akron and Cleveland, and the State of Ohio (Table 3). During this period average per capita personal consumption increased $4,504, or 23%. Contributing to the declines in per capita GPI were increasing costs of consumer durables (average county deduction of $1,794), increasing income inequality in many geographic areas (average -$1,324), increasing costs of leisure time loss (average -$973) and climate change (average -$777), declines in the value of household labor (average -$673), rising costs of nonrenewable resource depletion (average -$516), commuting (average -$214), and unemployment and underemployment (average -$106). Slight declines were also seen in the value of volunteer work, while increases were seen in the cost of household pollution abatement, noise pollution, and wetland and farmland loss, though the magnitude of these changes was not large relative to the overall GPI (Figure 4, Table 4).

Table 3: GPI per capita, 1990-2005

Geographic area	Per capita GPI, 1990	Per capita GPI, 2000	Per capita GPI, 2005	% change, 1990-2000	% change, 2000-2005	% change, 1990-2005
Ashland	$14,680	$15,421	$15,484	5%	0%	5%
Ashtabula	$13,700	$12,923	$13,942	-6%	8%	2%
Columbiana	$14,383	$14,622	$14,874	2%	2%	3%
Cuyahoga	$16,917	$15,272	$16,274	-10%	7%	-4%
Erie	$12,426	$11,372	$13,689	-8%	20%	10%
Geauga	$21,244	$22,136	$24,613	4%	11%	16%
Huron	$15,915	$14,034	$12,853	-12%	-8%	-19%
Lake	$17,327	$14,708	$13,041	-15%	-11%	-25%
Lorain	$17,709	$16,013	$16,037	-10%	0%	-9%
Mahoning	$14,767	$12,785	$14,748	-13%	15%	0%
Medina	$21,238	$19,840	$22,213	-7%	12%	5%
Portage	$15,985	$14,346	$16,387	-10%	14%	3%
Richland	$14,621	$14,512	$13,489	-1%	-7%	-8%
Stark	$16,245	$14,410	$15,818	-11%	10%	-3%
Summit	$16,036	$15,316	$16,081	-4%	5%	0%
Trumbull	$15,495	$12,687	$14,682	-18%	16%	-5%
Wayne	$14,664	$13,469	$14,335	-8%	6%	-2%
Akron	$13,781	$14,254	$12,826	3%	-10%	-7%
Cleveland	$10,213	$11,034	$9,500	8%	-14%	-7%
Ohio	$16,993	$17,523	$16,855	3%	-4%	-1%
U.S.	$14,978	$15,198	$15,262	1%	0%	2%
Urban avg.	$13,637	$13,520	$12,867	-1%	-5%	-6%
Suburban avg.	$17,338	$15,805	$17,069	-9%	8%	-2%
Rural avg.	$14,341	$13,765	$14,095	-4%	2%	-2%

Table 4: Components of GPI change from 1990-2005 (average of 17 Northeast Ohio counties)

GPI component	Average county change, 1990-2005	Absolute value of change	Percent of total change
Personal Consumption Expenditures	$4,504	$4,504	35.6%
Adjustment for Income Inequality	-$1,324	$1,324	10.5%
Household Labor	-$673	$673	5.3%
Volunteer Labor	-$3	$3	0.0%
Household Capital	$384	$384	3.0%
Streets & Highways	$128	$128	1.0%
Crime	$7	$7	0.1%
Family Breakdown	$34	$34	0.3%
Leisure Loss	-$973	$973	7.7%
Unemployment & Underemployment	-$106	$106	0.8%
Consumer Durables	-$1,794	$1,794	14.2%
Commuting	-$214	$214	1.7%
Pollution Abatement	-$34	$34	0.3%
Vehicle Crashes	$206	$206	1.6%
Water Pollution	$2	$2	0.0%
Air Pollution	$38	$38	0.3%
Noise Pollution	-$1	$1	0.0%
Wetland Loss	-$8	$8	0.1%
Farmland Loss	-$34	$34	0.3%
Nonrenewable Resource Depletion	-$516	$516	4.1%
Climate Change	-$777	$777	6.1%
Ozone Depletion	$43	$43	0.3%
Forest Loss	$0	$0	0.0%
Net Capital Investment	$834	$834	6.6%
Genuine Progress Indicator	-$282	$12,641	

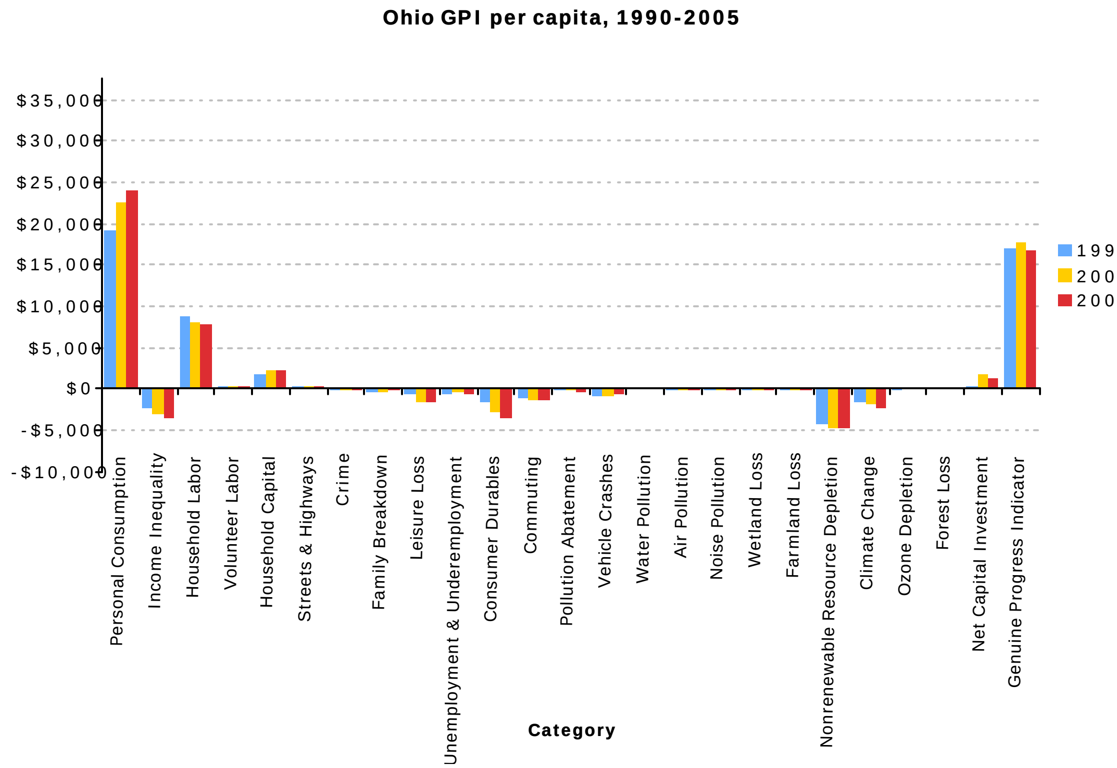


Figure 4: Change in GPI components from 1990-2005 for Ohio

Some positive contributions to GPI grew from 1990-2005. These included an increase in net capital investment (average +$834, derived solely from national data), services of household capital (average +$384) and highways and streets (average +$128), and a decline in the costs of vehicle crashes (average +$206). Slight reductions were also seen in the costs of crime, family breakdown, water and air pollution, and ozone depletion. Forest re-growth added value to some Northeast Ohio counties but forest loss deducted value in others. These added values, however, were not enough to overcome the increasing costs components. Average per capita GPI for

Ohio, the 17 counties, and cities of Akron and Cleveland declined by $504 between 1990 and 2005.

Examining the 1990-2000 and 2000-2005 time periods separately, we found generally greater per capita growth from 2000-2005. From 1990-2000, per capita GPI rose in 3 counties, the cities of Akron and Cleveland, and State of Ohio, and fell in 14 counties. From 2000-2005, per capita GPI rose in 14 counties and declined in 3 counties, the cities of Akron and Cleveland, and State of Ohio. These trends were similarly influenced by the relative growth of inequality-adjusted personal consumption versus other environmental and social costs. In geographic areas that had relatively high personal consumption growth without rising inequality, per capita GPI generally grew. Geographic areas with slowly rising personal consumption or greatly expanding inequality saw declines in per capita GPI.

3.2 Spatial trends: intra-regional comparisons

As expected, GPI and its component costs and benefits vary greatly across the region. Per capita GPI was highest in suburban regions and lowest in urban areas, with rural regions intermediate (Figure 5). Wealthier suburban areas generally had the greatest personal consumption, an important driver of GPI trends. Urban areas had higher income inequality and costs of unemployment and underemployment, leading to lower GPI. Suburban areas had higher per capita costs for some environmental and social components, such as farmland loss and commuting, owing to high rates of land conversion and automotive dependence.

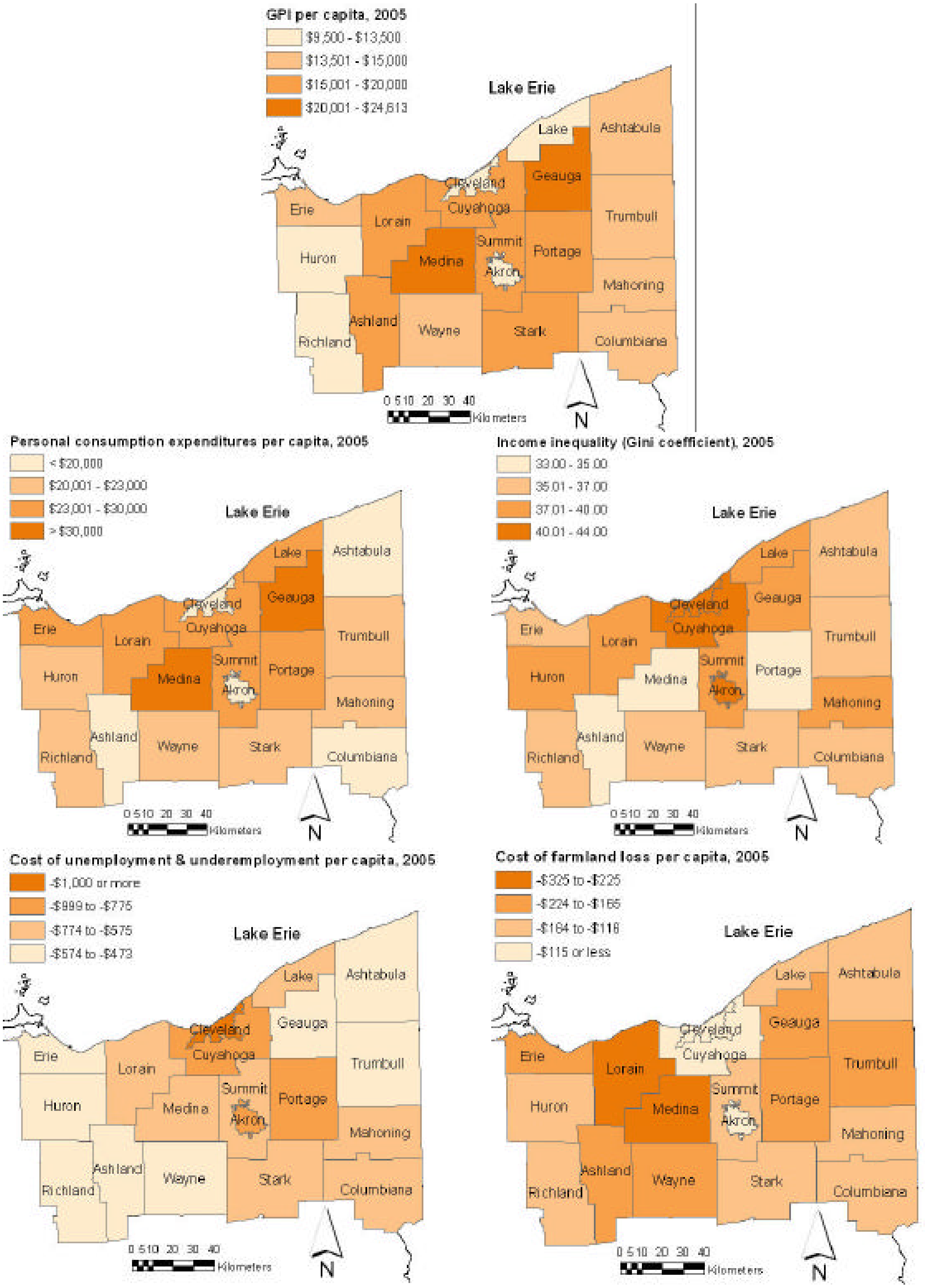
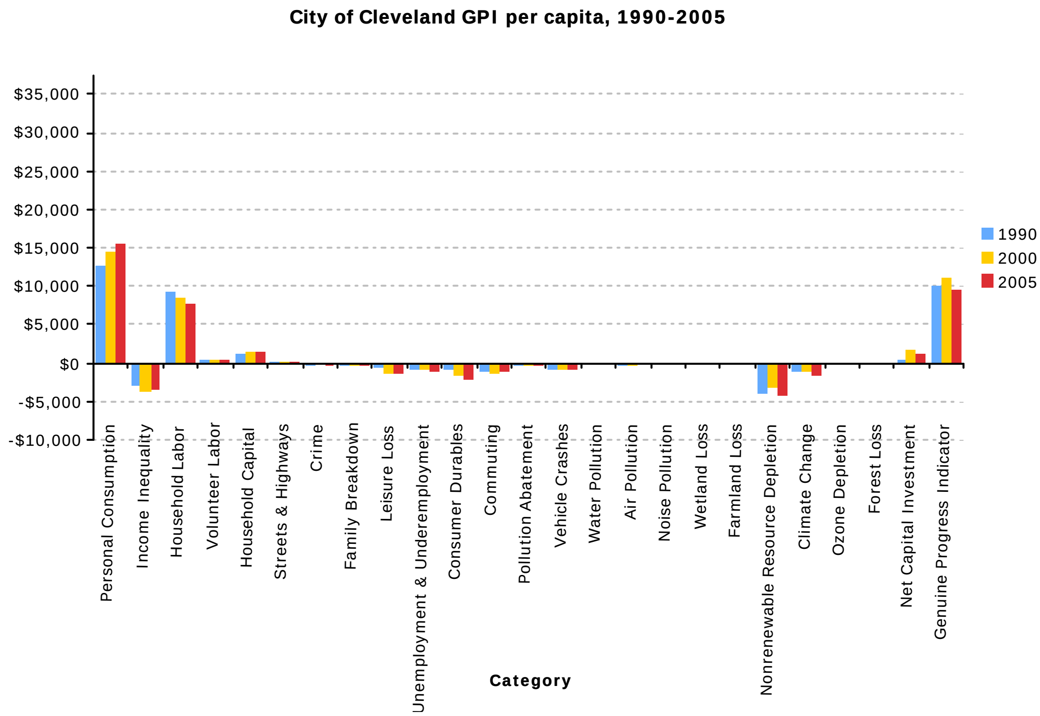


Figure 5: Differences in per capita GPI, personal consumption, income inequality, costs of unemployment and underemployment, and farmland loss across Northeast Ohio

An examination of all GPI components shows why GPI values diverge in different geographic areas (Figure 6). Geauga County, the wealthiest in the study area, saw the greatest percent growth in per capita GPI from 1990-2005 (Table 3). Geauga County did have high costs associated with suburbanization, including higher per capita costs of commuting, wetland and farmland loss, climate change, and nonrenewable resource depletion. However, these costs were more than offset by greater personal consumption, a consequence of larger per capita income. Cleveland, meanwhile, has low personal consumption, greater income inequality, and greater costs of crime and unemployment and underemployment. Huron and Lake counties saw the largest declines in per capita GPI from 1990-2005. Rising income inequality over this period was largely responsible for this decline, as inequality-adjusted personal consumption declined in these counties over the 15-year period. Huron and Lake counties were the only places to witness declines in inequality-adjusted consumption during this period.



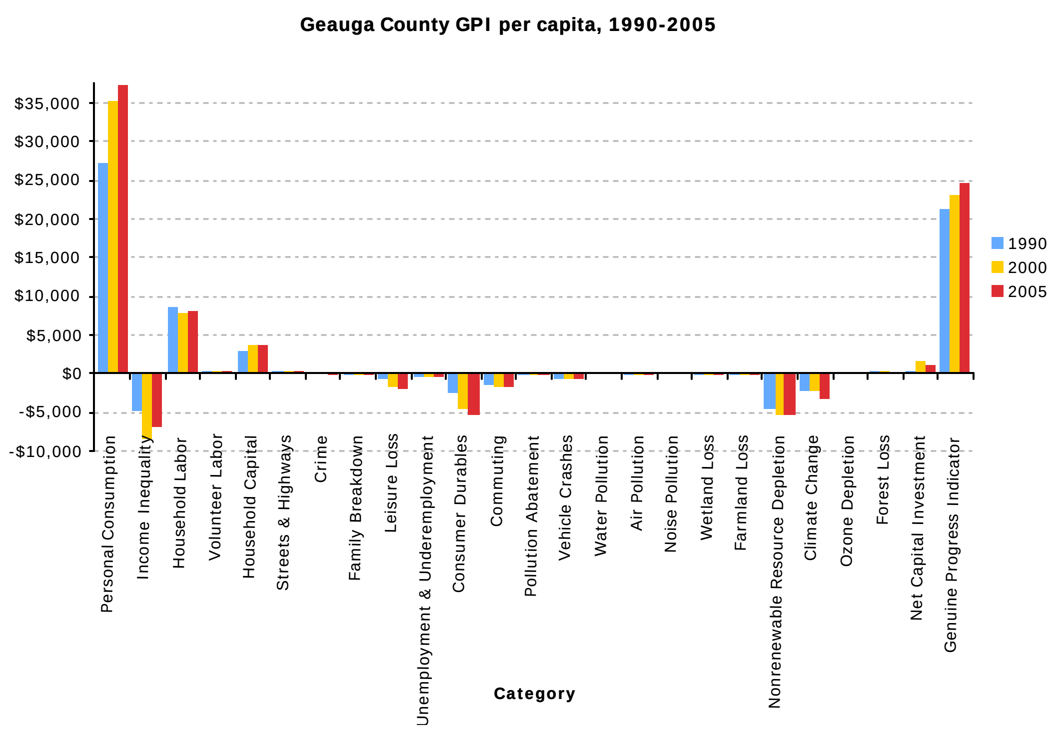


Figure 6: Change in GPI components from 1990-2005 for the City of Cleveland (top) and Geauga County (bottom)

3.3 Spatial trends: inter-regional comparisons

Per capita GPI is lower in Ohio than in corresponding Vermont geographic areas (i.e., for Ohio versus Vermont, rural Northeast Ohio versus rural Northern Vermont, and suburban Northeast Ohio versus Chittenden County, Vermont, Figure 7). Ohio and rural Northeast Ohio start with greater per capita personal consumption than Vermont or rural Northern Vermont. The primary contributors to Vermont's larger GPI are the smaller costs of climate change and nonrenewable resource depletion, and greater value of forest gain. Vermont geographic areas generally also had lower cost of vehicle crashes, consumer durables, commuting, and household pollution abatement, and greater value of volunteer labor and services of streets and highways. Ohio generally had greater value from services of household capital and less leisure time loss. However, these items totaled only 7% of the total adjustments to the GPI, and were thus small relative to the larger costs such as personal consumption (41%), income inequality (9%), household labor (14%), and nonrenewable resource depletion (9%).

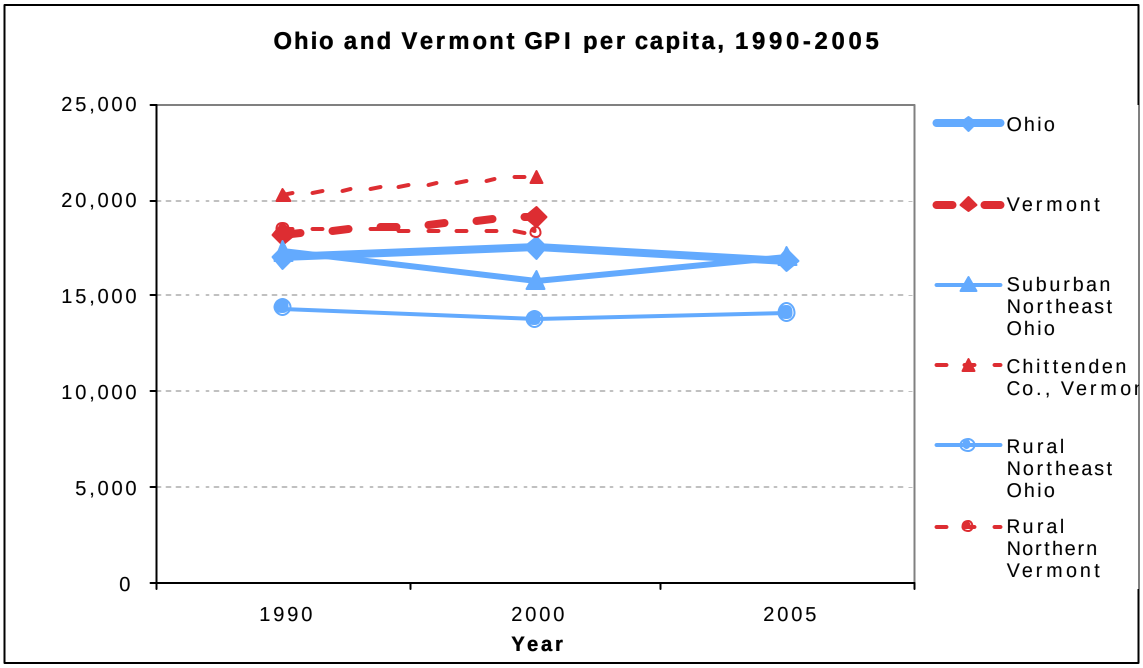


Figure 7: GPI per capita for corresponding Ohio and Vermont geographic areas, 1990-2005

Chittenden County, Vermont and Geauga County, Ohio illustrate two different paths to similarly high per capita GPI. These counties have the highest per capita GPI of their respective study areas. While Geauga County has greater per capita personal consumption and services of household capital, Chittenden County has lower income inequality, more value from household labor, and smaller costs of climate change and nonrenewable resource depletion (Figure 8).

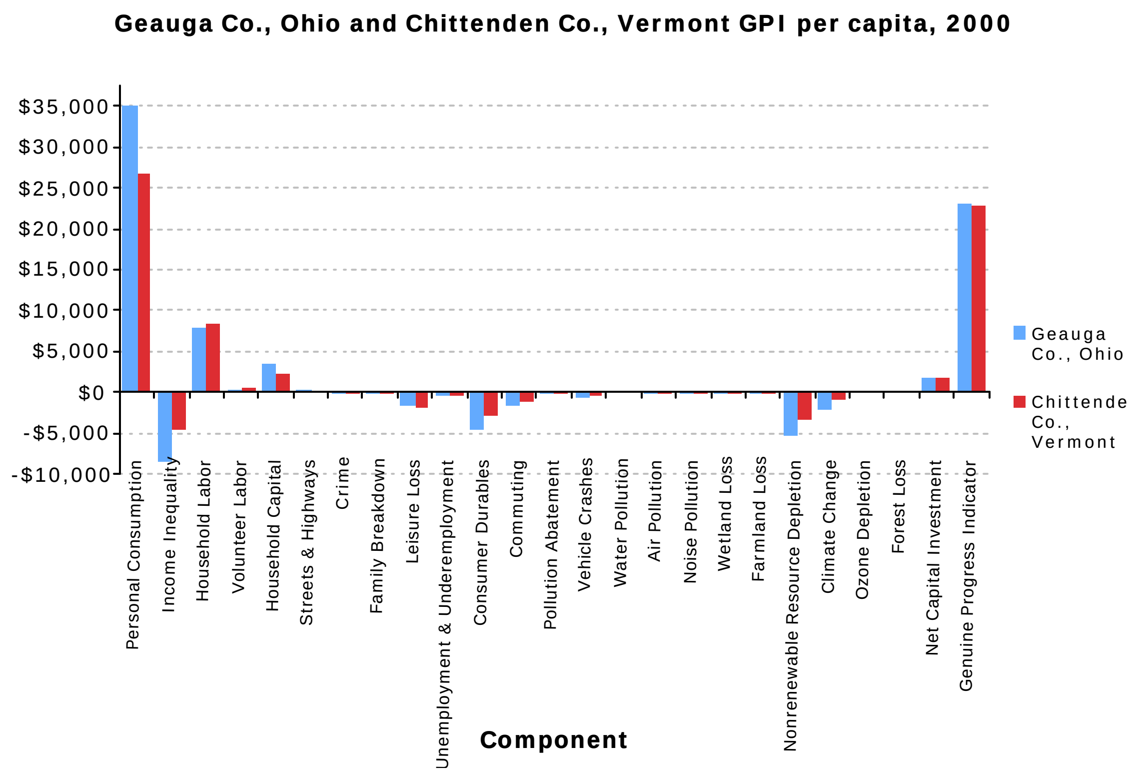


Figure 8: GPI per capita for Geauga County, Ohio and Chittenden County, Vermont for 2000

Per capita GPI in Ohio is similar to that in the U.S., with both U.S. and Ohio values below those of Vermont. However, since the national GPI was calculated using different methods (Talberth et al. 2007), state and national values are not strictly comparable.

3.4 Sensitivity analysis: Non-local data and non-conservative assumptions

Our baseline results presented above used local data wherever possible with generally conservative assumptions to measure and value economic, social, and environmental costs and benefits. Since the data and methods differ from past U.S.

GPI studies, we next compare the effects of these changes on our results. Absolute and relative changes should be addressed when data and methods are changed – first, how much do GPI values change and second, are the temporal and spatial trends between geographic areas preserved?

When local data were omitted, GPI values for all geographic areas changed, but not always equally. Without local data, GPI results were overestimated at the local level by as much as 22% and underestimated by as much as 23% in different geographic areas. GPI was overestimated in all years for Ashtabula, Cuyahoga, Erie, and Mahoning counties and the City of Cleveland, generally less wealthy geographic areas. GPI was underestimated in all years for Geauga, Huron, Lorain, Medina, Portage, Stark, and Wayne counties, generally wealthier geographic areas.

The omission of local data did not change spatial GPI rankings for most geographic areas. However the omission of local data did lead to relatively lower rankings for Ashland, Lorain, and Portage counties and relatively higher rankings for Cuyahoga, Erie, and Mahoning counties. Local data most responsible for absolute and relative changes in GPI trends included personal consumption, services of household capital/cost of consumer durables, the cost of underemployment and unemployment, and the cost of climate change. To be as accurate as possible, it is particularly important that future GPI studies obtain local data for these components.

Our "nonconservative assumptions" included:

1. Adding a social cost of murder along with the cost of property crime to the overall cost of crime (which has not been done in past GPI studies). Adding murder to

the cost of crime reduces per capita GPI in different geographic areas a maximum of 5%, but an average of less than 1%.

2. Using pre-settlement wetland cover as a baseline (as has been done in past GPI studies) rather than extrapolated wetland cover in 1940. Using a pre-settlement baseline for wetland loss reduces per capita GPI in different geographic areas up to 33%, with an average reduction of 5%.

3. Using replacement costs for biofuel for all nonrenewables depletion cost (as has been done in past GPI studies) rather than using solar and wind as replacements for electricity generation. The less realistic assumption that biofuels be substituted for all nonrenewable energy sources reduces per capita GPI in different geographic areas as much as 41% and an average of 13%.

4. Using greenhouse gas emissions data based on emissions from each sector (as has been done in past GPI studies) rather than based on greenhouse gas intensities of various consumer expenditure categories. The choice of which climate change cost measure to apply can lead to a change of up to 7% in GPI in different geographic areas, though the average change is near zero. Using consumer expenditures as a basis for calculating carbon emissions leads to higher costs in counties like Medina and Geauga, which had the greatest levels of personal consumption.

5. Accumulating the cost of ozone depletion over time (as has been done in past GPI studies). Accumulating the costs of ozone depleting chemicals reduces per

capita GPI in different geographic areas by an average of 7% and a maximum of 17%.

6. Using pre-settlement forest cover as a baseline (as has been done in past GPI studies) rather than 1940 forest cover. Using a pre-settlement baseline for forest loss reduces per capita GPI in different geographic areas up to 6%, with an average reduction of 2%.

Combining these non-conservative assumptions, GPI is reduced an average of 27%, a maximum of 66%, and a minimum of 10% for different geographic areas in different years (Figure 9).

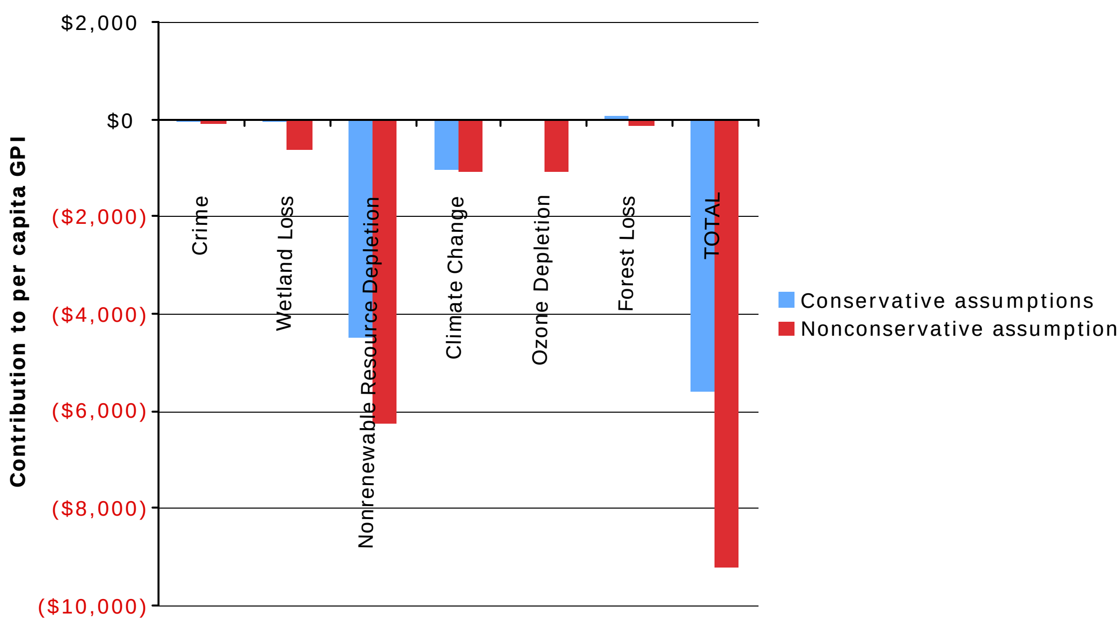


Figure 9: Effects of conservative vs. nonconservative assumptions on per capita GPI

4. Discussion

4.1 Temporal and spatial trends

The declines in per capita GPI for parts of Northeast Ohio from 1990-2005 are not unique, as authors of GPI studies have often documented stable or declining per capita GPI in recent years (Jackson and Stymne 1996, Lawn and Clarke 2008). However, per capita GPI grew in eight of 17 counties from 1990-2005. Aside from the fact that all urban regions declined in per capita GPI over this period, there is no clear spatial pattern to GPI's rise or decline. In the rural and suburban parts of Northeast Ohio, some rural and suburban counties' per capita GPI grew while others declined. Per capita GPI declined for most geographic areas from 1990-2000 while growing from 2000-2005.

The largest contributors to changing GPI were personal consumption, income inequality, household labor, leisure time loss, climate change and nonrenewable resource depletion, and net capital investment (Table 4). Changes to the cost of consumer durables and services of household capital were important but are closely related to levels of personal consumption expenditures. Where personal consumption grows faster than the various negative social, economic, and environmental costs, GPI grew. For example, Geauga, Erie, and Portage counties saw per capita personal consumption grow over 30% from 1990 to 2005. These counties all saw per capita GPI rise during this period. By contrast, Huron and Trumbull counties had the smallest growth in personal consumption, only 11%. These counties both had declines in per capita GPI from 1990-2005, particularly in Huron County, where income inequality also increased over this period. Places where personal consumption starts at a low level may also be unable to overcome growing social, economic, and environmental costs through growth in consumption. Despite 22% growth in per capita personal consumption and a decline in inequality, Cleveland, which started with the region's lowest personal consumption, witnessed a 7% decline in per capita GPI from 1990-2005.

As with intra-regional comparisons, the discrepancy between personal consumption and other environmental, social, and economic costs and benefits can explain differences in per capita GPI between Northeast Ohio and Northern Vermont. Ohio had greater per capita personal consumption than Vermont, as did rural Northeast Ohio versus rural Northern Vermont. However, the lower income inequality, greater value of forest re-growth, and smaller cost of climate change and nonrenewable resource depletion in Vermont typically led to greater per capita GPI than corresponding Ohio geographic areas. This pattern was also seen in Chittenden County, Vermont, which had similar environmental and socioeconomic performance as the rest of Vermont, combined with high levels of personal consumption. As such Chittenden County had per capita GPI nearly equal to that of Geauga County, Ohio, which had the highest GPI of any geographic area in Northern Vermont or Northeast Ohio.

Although Ohio's per capita GPI was smaller than Vermont's, such differences should be expected due to their radically different socioeconomic and environmental settings. Politically and economically, observers have considered Ohio to be reflective of the U.S. as a whole, a microcosm of national-scale political, socioeconomic, and environmental trends (Cleveland Plain Dealer 2004). Vermont, however, has a much different economic and demographic profile than much of the rest of the U.S., owing largely to its overwhelmingly rural geography. One key difference between these states is the source of electricity, which impacts climate change and nonrenewable resource depletion costs. In Ohio, where nearly 90% of electricity is generated using coal, these costs are greater than Vermont, which obtains 75-80% of its electricity from hydroelectric and nuclear power. Just because Vermont has higher GPI than Ohio does not mean that both states could not improve certain GPI component indicators. It also does not necessarily mean that it would be feasible or desirable for Ohio to adopt policies to improve well-being based on those in place in Vermont.

Due to changes in how the U.S. Bureau of Economic Analysis computes subnational GDP, it is difficult to construct recent time series of state-level per capita GDP. BEA (2003) provides a time series of state-level GDP from 1977-2001. These data show that Ohio's per capita GDP was typically greater than Vermont's from 1977-2001. However, Ohio's per capita GPI was consistently lower than Vermont's. When per capita GDP and GPI are indexed to 1980, Vermont's per capita GDP is found to have grown 51% from 1980-2001, while Ohio's grew 33%. However, Vermont's per capita GPI grew only 15% from 1980-2000, while Ohio's grew by 8% from 1980-2000 and 4% from 1980-2005 (Figure 9). These trends suggest that not all economic growth is created equally, and that the growth occurring in Ohio in recent decades has not supported well-being as measured by the GPI.

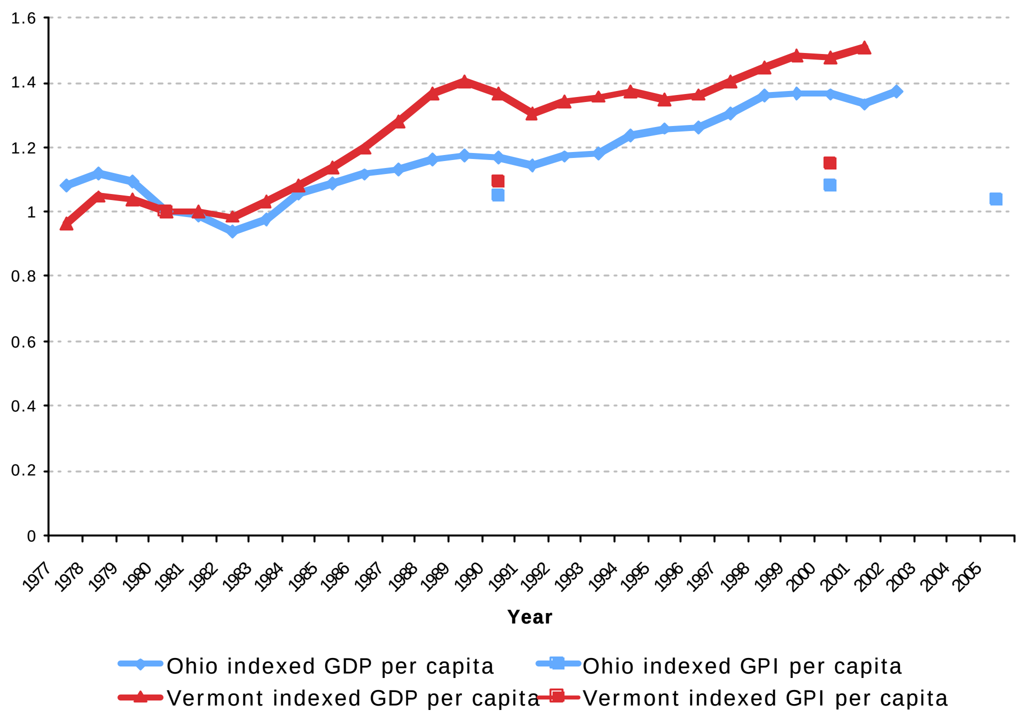


Figure 10: Indexed GDP and GPI per capita, Ohio and Vermont (1980 = 1)

4.2 Sensitivity analysis

Local data is clearly important to improving the accuracy of GPI estimates at the state, county, or city level. When state or national data are scaled down, absolute and relative trends may be compromised. Local GPI studies in the U.S. have used progressively higher quality local data, with this study using the most yet (Minnesota Planning Environmental Quality Board 2000, Costanza et al. 2004, Venetoulis and Cobb 2004, Bagstad and Ceroni 2007). Our choice to abandon serious attempts at analysis prior to 1990 was an important compromise on data quality. GPI estimates for these early years are of poor quality and depend on increasingly unreliable extrapolation for earlier decades. Going forward, having more high-quality GPI analyses that use good quality data to facilitate regional comparisons can help provide motivation for new jurisdictions to follow with their own studies.

Despite these gains, there is still room for improvement in local estimates. In particular, GPI components measuring time use rely on state-level data, with local estimates frequently not existing. Time use data influence estimates of household and volunteer labor, leisure time loss, and the cost of unemployment and underemployment. Improved local data for these categories would further benefit local GPI estimates.

With the exception of adding the cost of murder, which more comprehensively treats the cost of violent crime, we generally prefer to use conservative approaches to valuing economic, social, and environmental costs and benefits in the GPI. We felt it was important to use realistic and policy-relevant baselines for the cost of wetland and forest loss, since a return to pre-settlement land cover conditions is neither a realistic nor desirable public policy goal. Similarly we did not accumulate the cost of greenhouse gases and ozone depleting chemicals, attributing costs in a given year only to release in that year. There are certainly social legacy costs to past production and release of these substances. But for ozone depletion, we felt that the phase-out of CFC production by the Montreal Protocol means that current and future social welfare measurements should not be burdened by the costs of past actions. Finally, we used more realistic costs to account for the replacement of nonrenewable energy sources by renewable energy and to account for the carbon intensity of consumer spending rather than direct energy consumption. These methods improve local estimates while addressing the import-export problems of the GPI, at least for greenhouse gas emissions of imported goods or energy (Clarke 2007).

The large reductions in per capita GPI, as high as 66%, when using non-conservative assumptions are a potential source of much of the criticism behind use of the GPI to pinpoint when economic growth becomes “uneconomic” (Neumayer 1999, 2000). Since the author of a given GPI study effectively defines its system boundaries, critics of the GPI have argued that researchers are ideologically biased in the search for a measure that supports the threshold hypothesis. We feel that by including conservative but comprehensive assumptions and valuation methods, such ideological pitfalls can be better avoided.

4.3 Strong sustainability and the costs of urban decline and decentralization

Neumayer (1999) notes that GPI is not a measure of strong sustainability. It is entirely possible to deplete natural or social capital while expanding income and consumption to produce rising per capita GPI. This trend is observed in the wealthier suburbanizing counties in Northeast Ohio. Many of the externalities associated with suburban development, such as rising commute times, air and water pollution, and open space loss are monetized as part of the GPI. However, these external costs are often offset by increased wealth and consumption in these communities, which can lead to greater per capita GPI. By extending our treatment of the costs of climate change to include personal consumption expenditures, we do incorporate some of the social costs of increasing consumption. Similar to our results, Venetoulis and Cobb (2004) found that counties in the San Francisco Bay Area with the highest personal consumption per capita, Marin and San Mateo counties, also had the greatest GPI per capita. Alameda and Solano counties had the lowest per capita personal consumption levels and also the lowest per capita GPI.

Conversely, GPI seems to better capture the costs of urban decline. Cleveland and Akron both registered low per capita GPI due in part to low income, high inequality, and high costs of crime, unemployment, and underemployment. As such, GPI serves as a sometimes-imperfect measure of social well-being in the regional context. While critics have pointed out the strong sustainability problem with the GPI in theory, this study is the first to show how it operates in practice, by enabling comparisons of multiple geographic areas using the same methods.

Unfortunately, we were unable to reliably capture the long-term GPI trends during the period of suburbanization and urban decline that took place in the last half of the twentieth century. Local data from the 1950s-1980s were generally of poor quality, requiring extrapolation of present-day trends. Such extrapolation assumes that present-day socioeconomic and environmental trends held true in earlier decades. This assumption is likely invalid for the areas that have undergone the most change –urban centers, which have suffered population and employment loss, and suburban counties, many of which have grown from rural towns to today's suburbia.

4.4 Implications for future local GPI studies

Discussions on green accounting in the U.S. at the federal level have been frozen since the mid-1990s (Cobb et al. 1995). Thus community-level engagement with local and state decision makers based on the GPI may be a useful way to move this debate forward. Additional local and regional studies can also facilitate more interregional comparisons, as demonstrated in this paper for Ohio and Vermont. To enable comparisons, new studies should use comparable data sources and methods. In this study we have endeavored to develop and document methods that will enable accurate, policy-relevant measurements of the GPI for states, counties, and large cities. This can enable a better understanding of how the costs and benefits of economic growth are distributed within a nation. As described in the next section, local GPI estimates can also be used to engage local groups in discussions about regional sustainability and economic development.

5. The policy process

5.1 Well-being in Northeast Ohio

As residents of a region that has seen widespread loss of its industrial employment base, Northeast Ohioans have long been concerned with the state of the regional economy, particularly on maintaining employment opportunities. At the same time, interest has grown in building a region that takes more sustainable approaches to issues like land use, energy, and food systems. Organizations like Green City Blue Lake (GCBL, http://www.gcbl.org) have promoted both causes. Yet conventional economic measures like the region's "Dashboard of Economic Indicators" (http://www.futurefundneo.org/page10474.cfm) focus on economic indicators, along with a few social metrics. The Dashboard ranks Northeast Ohio's performance on indicators of economic growth against other urban areas, with the underlying assumption that further economic growth will provide more jobs, a primary social concern in the region. Yet if the region focuses solely on quantitative economic and employment growth, it may see declines in other aspects of regional well-being. For instance, the construction boom of the 1990s and early 2000s provided employment opportunities, but at the cost of open space loss, increasing traffic and automotive dependence, and loss of vitality in the region's urban centers. Northeast Ohio has recently witnessed controversy over payday lending and casino gambling, both potential employment sources with accompanying social costs. Job growth that provides external economic, social, and environmental benefits is more likely to improve the region's well-being than growth in sectors that erode social and natural capital.

Measures that are designed to account for economic, social, and environmental performance, like the GPI, offer one way to bridge the gap between these two views of the regional economy. The GPI to date has been primarily an academic exercise conducted to show both the benefits and costs of economic growth. However, members of the local academic and nonprofit communities are developing and implementing an outreach program based on the GPI. We hope this program can serve as a model for other communities interested in focusing on the broader impacts social impacts of economic decisions.

5.2 Moving beyond academia

To move beyond an academic exercise, GPI must obtain the popular support of policymakers and their technical staff. Haggart (2000) notes that "Government support is a major reason why the GDP was accepted, becoming the most widely used indicator. Only government can give an indicator program the recognition, the resources and the data base needed to make an indicator anything more than a semi-authoritative number designed to fit the needs – ideological, financial or otherwise – of its creator." As such, GCBL and Oberlin College are collaborating on a series of workshops to build support for GPI-based indicators in Northeast Ohio. As a local nonprofit institution, GCBL's goals include making results and methods more accessible to the public and local decision makers. By collaborating with academic institutions, nonprofits like GCBL can provide publicity, host workshops and training events, and provide a "home" for periodic updates of the region's GPI estimates.

5.3 Framing the issues

GPI trends for much of Northeast Ohio are quite similar to national trends, with some areas gaining and some losing but the region as a whole remaining flat or declining slightly. From a policy perspective, it is important not to frame these trends as a cause for hopelessness, but as a set of indicators that can be improved upon at the local and regional level. Local governments have little control over monetary and fiscal policy, or national and global business cycles. However, they can make strategic choices about education, infrastructure, land use, energy, and other types of investment. GPI could also promote regionalism by demonstrating that gains in certain areas brought about by intra-regional migration register as losses in other areas, while the entire region's well-being remains flat. Adoption of a qualitative economic development mentality, versus a quantitative growth mentality is typically politically difficult. However, Northeast Ohio presents an interesting opportunity for examining alternative regional development strategies, for two reasons. First, due to the job loss of recent decades, a blind rush toward economic globalization has not been universally accepted as desirable. Second, many leaders realize that in order to attract a creative, skilled work force, it is increasingly necessary to protect and enhance "quality of life" amenities in the region. If the GPI can be measured regularly and be indicative of changes brought about by these policies, it has the potential to serve as a useful local indicator in guiding policy.

Acknowledgements

Data collection, preparation, and calculations for many of the GPI components were completed by Greta Bradford, Colin Miller, Carl Schubert, and the students of the spring 2008 ENVS 220 course at Oberlin College. Partial support for this project was provided by Oberlin College and the University of Vermont. Brent Bleys, Steve Posner, and John Talberth, provided helpful suggestions on methods. David Beach,

Brad Chase, and Laura Christie at Green City Blue Lake assisted in developing and implementing the GPI outreach program for Northeast Ohio.

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CHAPTER 5: CONTEXT MATTERS: APPLYING ECOLOGICAL AND SOCIOECONOMIC CRITERIA FOR IMPROVED ECOSYSTEM SERVICES BENEFITS TRANSFER[1]

Abstract

Value transfer is an approach for applying economic values in policy contexts where those values were estimated at a different site. It has emerged as an important tool for ecosystem service valuation, due to its speed and ability to deal with incomplete local data. Yet the failure of many value transfer studies to account for landscape scale ecological and socioeconomic heterogeneity can reduce their accuracy, theoretical value, and policy value. We explored the benefits of using flexible, more descriptive land use land cover (LULC) typologies incorporating contextually important ecological and socioeconomic variables, in contrast to the less information-rich typologies typically used in value transfer. We reviewed past ecosystem service meta-analysis studies to identify instances where authors have noted the importance of such variables. Past authors have already validated the importance of many of these variables, but have not yet developed a systematic way to describe, catalog, and use them. We then identified cases within a recent value transfer for New Jersey where ecological and socioeconomic variables can be used to create a more flexible and descriptive LULC typology. We compared the range in ecosystem service values

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[1] Bagstad, K.J., A. Troy, and S. Liu. In preparation. Context matters: Applying ecological and socioeconomic criteria for improved ecosystem services benefits transfer. Target journal: Environmental and Resource Economics.

between the original and descriptive typologies. A more descriptive typology narrowed value ranges by 3-37% for “high value” LULC types and 60-93% for “low value” LULC types, versus the original typology. Systematic reporting of well-defined contextual variables in primary studies will improve ecosystem services meta-analyses, function transfer, and value transfer applications. Such improvements would advance both the theory and practice of ecosystem services valuation.

Keywords

Ecosystem services, value transfer, benefit transfer, land use/land cover classification, mapping, valuation, meta-analysis, management

1. Introduction

The quantification of ecosystem services is a field of growing interest to a wide range of conservation professionals, researchers, and policy advocates. Ecosystem services are commonly defined as the benefits humans derive from the structure and processes occurring in naturally functioning ecosystems, and are often quantified monetarily. They are typically conceptualized as an annual flow of goods and services derived from a stock of “natural capital” capable of generating this flow of services (Daly and Farley 2003). Successful policy application of ecosystem services can improve decision making for conservation finance, promote more sustainable use of public and private lands to provide increasingly scarce environmental public goods, and promote social equity (Costanza et al. 1997, Daily 1997, Balmford et al. 2002, Farber et al. 2002, Banzhaf and Boyd 2005, National Research Council 2005, Farber et al. 2006). Diverse stakeholders have an interest in better understanding the nonmarket economic values provided by natural capital – from land managers interested in comparing management strategies, to NGOs advocating for stronger conservation finance and greater protection for nature, to decision makers weighing the tradeoffs between land use decisions, to academics that ideally can help inform these processes.

Successful ecosystem services management requires an understanding of the spatially heterogeneous distribution of ecosystems across the landscape (Troy and Wilson 2006, Ruhl et al. 2007, Fisher et al. 2009). This understanding has historically been difficult to achieve. Ecological and socioeconomic systems are inherently complex and unpredictable, which is one reason that ecosystem services are not easy to map, assess, and value (Limburg et al. 2002). Valuation studies are costly and time-consuming and can rarely be prepared in the time frame required for policy applications. Given these difficulties, the practice of value transfer (Brookshire and Neill 1992, Wilson and Hoehn 2006) has grown in popularity to speed the mapping and valuation process. Value transfer uses economic values from studies estimated previously at a study site and then applies these values to a policy site.

Despite the promise of value transfer, it is still a relatively new process. The vast majority of value transfer studies appear in the gray literature. Given that these studies are often directed toward real-world policy applications, this is not overly surprising. In the peer-reviewed literature, Velarde et al. (2005) and Ingraham and Foster (2008) provide two of the few original, peer-reviewed value transfer exercises.

Kreuter et al. (2001) use ecosystem service values derived by Costanza et al. (1997) to estimate change in ecosystem services due to land cover change near San Antonio, Texas. Several other unpublished studies also simply apply Costanza et al.'s (1997) per-acre values to the land cover types present at the site of interest. Such approaches introduce large margins of error by using globally estimated numbers based on studies and methods that are considered increasingly dated. In the non-peer reviewed literature, some authors take shortcuts and do not rigorously applying the criteria needed for valid benefits transfer. Several authors have described the criteria needed for sound value transfer, including basic equivalence of the population, institutional setting, environmental resource, and constructed market characteristics (Boyle and Bergstrom 1992, Desvousges et al. 1992, Brouwer 2000, Spash and Vatn 2006, Plummer 2009).

There is clear danger in transferring values between study and policy sites without considering the similarity of ecological and socioeconomic contextual factors. For example, Breaux et al. (1995) obtain an extremely high per-acre value ($34,700/ac net present value) for wetlands removing nutrients from effluent at a small potato chip factory in southern Louisiana. To extrapolate this value over expansive wetland areas or to a watershed receiving no anthropogenic effluents, however, would clearly overestimate such values. Similarly, ecosystem service values can be inappropriately underestimated using value transfer. For example, it would be inappropriate to transfer Haener and Adamowicz' (2000) relatively low recreational value ($0.14/ac-yr) from Alberta's vast boreal forest to an urban setting, where natural capital is far scarcer and usually more valuable. Function transfer, rather than point transfer, (Loomis 1992) is designed to reduce these potential misapplications by applying a mathematical function that accounts for differences in resource characteristics, geographic setting, and the constructed market. Function transfer is preferable to point transfer, where values from a study site are transferred to a policy site with no adjustment. Yet to date, function transfer has been used relatively sparsely. In one example, Schuyt and Brander (2004) attempted to estimate the value of the world's wetlands using a meta-regression developed for wetland values. Regrettably wetlands are one of the few cover types with enough primary studies to conduct a rigorous meta-regression. The lack of primary valuation studies, which leads to a shortage of quality meta-analyses, limits the opportunities to use function transfer for other ecosystem types.

One of the first and most basic steps in spatially explicit value transfer is the development of a typology for land use/land cover (LULC, inclusive of both land and open water cover), which can then be related to ecosystem service flows for that LULC type (Troy and Wilson 2006). Unfortunately typologies are typically *ad hoc,* subjective, and imprecise, and hence different typologies are generally inconsistent and incompatible. In "real world" value transfer applications, land cover typologies are often based largely on data availability. In this case, the classification system may be arbitrary and poorly suited to ecosystem service analysis. Alternatively, authors may develop typologies that focus on a particular ecosystem service or ecological characteristic of interest (e.g., habitat for a certain endangered species or flood control benefits at the watershed scale). Typologies informed by this approach are at least based on locally important issues, but *a priori* assumptions may cause important services to be overlooked. Further, each instance of a typology of this type must be individually tailored to the local context, so adapting it to other contexts may prove awkward or impossible.

Finally, different typologies may use words inconsistently because of semantic ambiguity or subjectivity. This problem has become more apparent with the increasing availability of high resolution, high precision data that allow for very fine distinctions. For instance, the term “forest” has an intuitive meaning to many. Yet during a succesional transition from abandoned field to shrubland to immature forest, there is less consensus on the precise threshold where the area would be classified as “forest.” Similarly ecologists in different regions may have varied thresholds for defining grasslands, savannas, woodlands, and forests. The same issue applies to contextual factors. For instance, what researchers consider to be an “urban forest” may vary considerably, based on differing standards for population density, urban proximity, and patch size.

Commonly used LULC classification systems dating to Anderson et al. (1976) and its successors (e.g., the USGS National Land Cover Database, NLCD) are poorly suited to ecosystem services mapping, assessment, and valuation. There are at least four reasons why these systems work poorly for ecosystem services analysis: 1) most map land cover, rather than land use, and land use is an important indicator of ecosystem function, since factors related to use may affect ecosystem service flows (e.g., agricultural and forestry practices, or institutional use of open space, Verburg et al. 2009). 2) The diverse and numerous influences on ecosystem service delivery do not lend themselves to rigid hierarchical classifications. 3) Most LULC classifications assume total homogeneity within mapping units (generally pixels); in combination with their rigid hierarchies, this makes them unsuitable for environments with micro-scale heterogeneity, yet this heterogeneity is often crucial to characterizing differences in ecosystem service flows. 4) Most importantly, these systems fail to account for contextual factors, both ecological and socioeconomic, that would be expected to affect the services provided by a particular land cover. One of the key problems is that there are simply too many potentially important contextual factors to be considered in a static and hierarchical LULC classification. Therefore, aside from noting urban density, most LULC systems completely ignore socioeconomic characteristics. Although some recent LULC systems consider the interactive effects of natural land cover, human land use, and human population characteristics (hence land use and population effects on ecosystem processes), these systems do not explicitly consider ecosystem services (Brenner et al. 2006, Ellis and Ramankutty 2008). Recent meta-analyses have demonstrated the importance of socioeconomic characteristics such as per capita income, population density, and urban proximity (Shrestha and Loomis 2003, Brander et al. 2006) for determining ecosystem service values, illustrating the need to incorporate these characteristics into LULC typologies for ecosystem services.

To date, there has been no attempt to develop a universal LULC typology that is geared toward the characterization of ecosystem services. One potential approach would be to use a relational rather than hierarchical system. This would entail a predefined set of broad base LULC classes (i.e., "forest" or "grassland") with further description added by using an expandable set of contextual variables to describe ecological and socioeconomic conditions. These contextual variables could be used to describe different dimensions of a given unit of land cover, such as the characteristics of its beneficiaries, its proximity to beneficiaries, its scarcity, or its internal site characteristics. Any particular combination of contextual variables could be applied to a specific mapping unit, which would be presumed to deliver a similar "basket" of ecosystem services associated with that modified class. While contextual variable definitions would be standardized, the variables themselves would be adaptable and open to changes and contributions from the ecosystem services research community. Banzhaf and Boyd (2005) recognize the importance of such contextual variables, which Boyd and Wainger (2002, 2003) term "landscape indicators." Banzhaf and Boyd (2005) classify these into three categories – socioeconomic characteristics of the user population, ecosystem "quality" factors (i.e., ecological characteristics of the natural system), and scarcity factors (including substitutes and compliments). Having such a predefined system would improve the flexibility, ease of use, and objectivity in LULC classification for ecosystem services assessment.

It is common for there to be an extremely wide range of value estimates for an ecosystem service produced by a given LULC type. A wide value range renders the use of average values as less meaningful, and reduces the utility of ecosystem services valuation for policy. Differences of several orders of magnitude are common, as are standard deviations that exceed the range of values. One possible reason for these discrepancies is that averages are taken across broad classes that include considerable heterogeneity (e.g., "forest"). These large variances could be reduced by creating more precisely defined classes (e.g., early successional hardwood forest, rather than "forest").

In this paper we explore the theoretical and practical possibility of using a flexible, descriptive, systematic LULC typology over traditional *ad hoc* typologies. We first reviewed past ecosystem services meta-analyses to identify instances where authors have already recognized theoretically and statistically important contextual variables. These variables could form the basis of a highly descriptive LULC typology for ecosystem services. Meta-analysis has been increasingly used over the past two decades to synthesize primary research, test hypotheses about the effects of explanatory variables, and develop meta-regressions for function transfer (Stanley 2001, Smith and Pattanyak 2002). We examined the multiple regression models developed in these studies to identify the influence of ecological and sociological contextual independent variables on ecosystem service value, the dependent variable.

Second, we reviewed a typical large value transfer exercise and compared the original typology developed during the study to a more descriptive typology that could be used for the same study by applying ecological and socioeconomic contextual variables to better define the context of the natural capital being valued. We hypothesized that the more descriptive typology will produce narrower ranges of value estimates than the original, *ad hoc* typology. Tightening the range of ecosystem service value estimates with a precise LULC typology would be practically and theoretically valuable. Such a system would both improve the validity of value transfer and provide more useful guidance to the scientists, practitioners, and policymakers interested in improved mapping, valuation, and accounting for ecosystem services.

2. Methods

2.1 Meta-analysis review

We first reviewed a series of past ecosystem services meta-analyses and cataloged the contextual variables that these authors either hypothesized or found to be statistically important influences on ecosystem service values. Relevant meta-analyses used multiple regression to analyze the effects of potential contextual variables on ecosystem service values (Smith and Kaoru 1990, Walsh et al. 1992, Sturtevant et al. 1995, Rosenberger and Loomis 2001, Woodward and Wui 2001, Brouwer et al. 2003, Brander et al. 2006, Brander et al. 2007, Ghermandi et al. 2008, Liu and Stern 2008). These papers estimated the effects of a variety of potential contextual variables on ecosystem service values, including income, urban proximity, population density, biodiversity, wetland type, and ecosystem area. We note however that the intent of these meta-analyses has not always been to identify ecological and socioeconomic contextual variables. Many of these studies test for differences between the value of specific ecosystem services, differences in valuation methods, or primary study quality. Regrettably, few authors of primary economic valuation studies comprehensively described the important contextual variables in their study system. As such, meta-analyses often tested only a small subset of potential contextual variables that are commonly reported in the literature or that are not overly time consuming for the meta-analyst to classify *post hoc*. Moeltner et al. (2007) refer to this as the meta-analyst's "n vs. k dilemma", where the researcher must choose between using more independent variables with fewer studies reporting them all or discarding potentially important variables but incorporating more studies.

2.2 Typology comparison

For our second approach, we explored the sensitivity of value estimates from a past value transfer study to changes in the LULC typology used. Many value transfer studies include both a low and high value estimate for a given ecosystem service, due to the uncertainties inherent in nonmarket valuation. We compared these high-low value ranges using the *ad hoc* typology initially developed for the study to those obtained by using a more descriptive typology developed using appropriate ecological and socioeconomic contextual variables. Due to the small sample size of primary studies available to estimate these value ranges, we compared the range of values rather than their standard deviation for each LULC type. After reviewing past value transfer exercises, we chose a study estimating ecosystem service values for the state of New Jersey (Costanza et al. 2006) for further analysis. Although this study is not published in the peer-reviewed literature, it does apply well defined selection criteria for studies used, develops its own land cover typology, and transparently reports a range of values. Like other value transfer studies, this example reports a wide range of value estimates for each land cover type. This is largely due to the uncertainty inherent in value transfer, as a range of values is typically reported rather than a single dollar value for each ecosystem service produced by a given LULC type.

When choosing a value transfer study to closely review, we faced several limitations. Many value transfer studies simply use ecosystem service value coefficients from sources like Costanza et al. (1997) (e.g., Kreuter et al. 2001). Several other studies (Herrera Environmental Consultants et al. 2004, TSS Consultants 2005, Brenner 2007, Swedeen and Pittman 2007, Ingraham and Foster 2008) that we reviewed used point transfers from a highly similar collection of primary studies as their basis for value (e.g., based on an EVRI search, using peer reviewed studies from the U.S., Canada, and western Europe). As such our results would likely be similar had we chosen any of these studies for analysis rather than that of Costanza et al. (2006).

To develop the more descriptive typology, we identified those ecosystem services produced by a given LULC type in the original typology whose high and low value estimates (in dollars/acre) differed by an order of magnitude or more. We explicitly avoided comparing values for carbon sequestration and storage, since most of the valuation differences in the literature for this service relate to how the value of carbon is measured (i.e., discount rate used, use of equity weighting, or assumptions about thresholds and catastrophic events, Tol 2005), as opposed to the characteristics of the natural capital itself. We then examined the primary studies used to estimate value for that service to identify differences in their ecological or socioeconomic setting. When evaluating studies for such differences, we compared only studies that used the same valuation method (e.g., contingent valuation, hedonic, travel-cost) to avoid mistaking differences between environmental resources with differences in methods. Fortunately similar methods are often used to measure the value of specific ecosystem services (e.g., hedonic pricing to measure aesthetic value, replacement/avoided cost to measure the value of regulating services, travel cost to measure recreational value). When we were able to identify a key ecological or socioeconomic difference between studies, we created a more descriptive LULC category based on that difference. As such, we could test the extent to which more descriptive LULC categories resulted in ecosystem service value estimates with narrower value ranges.

3. Results

3.1 Meta-analysis review

Although meta-analyses often include independent variables that are not ecological or socioeconomic contextual variables (e.g., ecosystem service valued, differences in primary study methods, characteristics of survey respondents), past authors have used at least 15 contextual variables as explanatory variables in meta-regressions (Table 1). These include seven ecological variables: biodiversity, ecosystem area, lake/river, coastal proximity, salinity, wetland type, and coastal ecosystem type, and nine socioeconomic variables, including "marquee status[2]," "environmental pressure[3]," ownership status, substitute sites, site amenities/development level, congestion, population density, income or GDP per

[2] We define "marquee sites" as those that are regionally, nationally, or internationally notable, e.g., national parks, World Heritage sites, or Ramsar wetlands.
[3] A weighted combination of wetland hydrologic alteration, proximity to urban or agricultural land use, and protected status (Ghermandi et al. 2008).

capita, and urban proximity[4]. These ecological variables are typically assumed to influence the ecosystem's ability to supply certain quantities of a given service, while socioeconomic variables describing user population characteristics and scarcity influence demand for ecosystem services. Additionally, meta-analyses often include a regional variable, usually a dummy variable separating one or more geographic regions under study.

[4] Though urban proximity is actually indicative of several modifiers that often lead to higher ecosystem service value: increased scarcity of natural capital, potentially increased income, greater number of users, and smaller physical area, resulting in greater per-acre values.

Table 1: Summary of meta-analysis review for contextual variables

Contextual variable	# of studies	Variables effects on ecosystem service value & studies
Biodiversity levels	1	Nonsignificant (coral reef recreation, Brander et al. 2007)
Ecosystem area	6	Significant positive (coral reef recreation, Brander et al. 2007, coastal ecosystem services, Liu and Stern 2008); non-significant (wetlands, Brouwer et al. 2003); significant negative (wetlands, Woodward and Wui 2001, Brander et al. 2006, Ghermandi et al. 2008)
Lake/river	4	Negative (lake & river recreation, Smith and Kaoru 1990); significant positive to nonsignificant (Sturtevant 1995); significant positive (rivers, Rosenberger and Loomis 2001, Shrestha and Loomis 2003)
Coastal proximity	1	Nonsignificant negative (wetlands, Woodward and Wui 2001)
Salinity	1	Nonsignificant (wetlands, Brouwer et al. 2003)
Wetland type	3	Effects of wetland type on value differed by study since different authors classified wetlands inconsistently (Brouwer et al. 2003, Brander et al. 2006, Ghermandi et al. 2008)
Coastal ecosystem type	1	Significant negative for beach, estuary, open ocean (coastal ecosystem services, Liu and Stern 2008)
Marquee status	6	Positive to negative (recreation on State/National parks, Smith and Kaoru 1990), positive, nonsignificant (National Forest recreation, Walsh 1992); negative, nonsignificant (National Forest recreation, Rosenberger and Loomis 2001, Shrestha and Loomis 2003); significant negative (wetlands, Brander et al. 2006); nonsignificant (coastal ecosystem services, Liu and Stern 2008)
Ownership status	2	Significant positive for public ownership (recreation, Rosenberger and Loomis 2001, Shrestha and Loomis 2003)
Substitute sites	3	Significant negative with more sites (recreation, Sturtevant et al. 1995, Rosenberger and Loomis 2001); significant negative (wetland abundance, Ghermandi et al. 2008)
"Environmental pressure"	1	Significant positive (wetlands, Ghermandi et al. 2008)
Site amenities/ development level	1	Significant negative (recreation, Shrestha and Loomis 2003)
Congestion	1	Significant negative (coral reef recreation, Brander et al. 2007)
Population density	4	Significant positive (wetlands, Brander et al. 2006, Ghermandi et al. 2008); nonsignificant (reef recreation, Brander et al. 2007); nonsignificant positive (coastal ecosystem services, Liu and Stern 2008)
GDP or income per capita	5	Significant positive (wetlands, Brander et al. 2006, Ghermandi et al. 2008); nonsignificant (reef recreation, Brander et al. 2007); nonsignificant positive (recreation, Shrestha and Loomis 2003); significant positive (coastal ecosystem services, Liu and Stern 2008)
Urban proximity	3	Significant positive (wetlands, Brander et al. 2006, Ghermandi et al. 2008); nonsignificant (coastal ecosystem services, Liu and Stern 2008)

It is difficult to draw broad comparisons about the significance of a particular variable between different meta-analyses. In some cases, contextual variables are inconsistently defined between studies (e.g., wetland type, geographic region). In other cases, meta-analyses differ regarding the significance and magnitude of the coefficient for the same independent variable. This is often the case when different ecosystem services are valued as the dependent variable. Nonsignificant values should not be taken as proof that a contextual variable does not affect the value of a service, however. Regression coefficients and their significance can differ based on several factors, including the functional form of the regression model, the number of studies available with that contextual variable, the studies actually selected for inclusion, the dependent variable used in the equation, and the interactive effects of independent variables. Lastly, important contextual variables may remain untested in meta-analyses since multiple primary studies (which often do not exist) are needed to evaluate the contribution of a given variable to a regression equation.

3.2 Typology comparison

The New Jersey value transfer evaluated the provision of 12 different ecosystem services across 13 LULC types, for a total of 156 ecosystem services produced by a given LULC type (or "ecosystem service-LULC type"). Of these 156 possible ecosystem service-LULC types, 35 types have economic values reported. Of these 35, thirteen ecosystem service-LULC types had value ranges that differed by an order of magnitude or more. The list of ecosystem service-LULC types with a wide value range is relatively short in part because of the limited number of primary studies on which value transfer studies draw. It is likely that as the literature grows, a similarly wide range will become apparent for other combinations. This range will likely be even larger if the resources valued are poorly defined in the primary literature. Of these 13 cases, eight ecosystem services-LULC types had contextual variables potentially responsible for the wide value ranges (Table 2). Seven socioeconomic variables (urban proximity, recreational vs. commercial use, pollution levels, rental vs. residential property, visitor use, substitutes, and per capita income) and one ecological variable (ecosystem area) were used to improve the original typology's precision. The fact that more socioeconomic variables were found to be potentially influential in improving the typology's precision may be due to poor description of the ecological resources in the primary valuation literature. We note that these eight cases are skewed toward aesthetic, recreational, and habitat services. This largely reflects the state of the primary literature on ecosystem service valuation, which is weighted heavily toward human preference-based econometric studies.

Table 2: Land cover-ecosystem service combinations with large value ranges potentially explained using contextual variables

LULC type	Ecosystem service					
	Climate regulation	Disturbance regulation	Waste treatment	Water supply	Habitat refugium	Aesthetic & recreational
Beach		X				RRP
Estuary					EA, PCI, RCU	EA, PCI
Forest					EA, UP	UP
Fresh wetland				X		X
Open fresh water						EA, NS, VU
Riparian buffer				PL, UP		X
Saltwater wetland			PPSP		X	
Urban open space	X					

X: Land cover-ecosystem service combination with high-low value range differing by an order of magnitude or more but not explainable using contextual variables

Contextual variable: Land cover-ecosystem service combination with value range potentially explained using contextual variables

EA: Ecosystem area
NS: Number of substitutes
PCI: Per capita income
PL: Pollution levels
PPSP: Proximity to point source pollution
RCU: Recreational vs. commercial use
RRP: Rental vs. residential property
UP: Urban proximity
VU: Visitor use

In the other five ecosystem service-LULC types, some value differences were due to variation in other factors, such as econometric methods used or choices of model specification. In others the source of value differences could not be determined because dependent variables did not facilitate comparison or the inclusion of too little ecological and/or socioeconomic information by the authors. Value estimates conducted in different time periods may also differ based on varying sophistication of methods, differing relative scarcity of natural capital in the landscape, and differing user population demographic characteristics, awareness, tastes, and preferences for environmental public goods.

Based on these eight contextual variables, we expanded the original *ad hoc* land cover typology for New Jersey to create new classes based on variables that better describe the likely causes of value differences in the original typology (Table 3). As expected, the value ranges are narrower for the more descriptive, systematic typology than the original *ad hoc* typology. Splitting the original typology into a more descriptive typology tended to produce “higher value” LULC classes that generally received greater human use, and hence had greater value, and “lower value” LULC classes receiving less use and of lower value. For the more descriptive LULC classes, value ranges are substantially tighter for “lower value” LULC classes than “higher value” LULC classes. The more descriptive typology greatly reduces the value range of lower value LULC types, by eliminating unreasonably high value estimates, while it does not greatly reduce the value range for higher value LULC types. For example, urban proximate forests have greater value, mainly due to their important recreational and aesthetic benefits combined with relative scarcity. By contrast, urban distant forests are less scarce on a per acre basis, and *ceteris paribus* receive relatively less recreational use. Thus urban distant forests, the lower value class, have a 68.4% narrower value range than forests, while urban proximate forests have only a 3.1% narrower value range than forests. Other “lower value” LULC types had value ranges 60.4-93% narrower than their corresponding original class, while other “higher value” LULC types had value ranges 6.6-37.1% narrower. For saltwater wetlands proximate to industrial point sources, the value range actually rose when using the more descriptive typology. This is because such wetlands may have a very high per-acre value, which was not used in the original study since it would have been too great a value to extrapolate to all saltwater wetlands. In this case using a more descriptive typology lets us more confidently use a higher value estimate once we can ensure that it is appropriately applied only to wetland areas likely to have such high values.

Table 3: Value ranges for New Jersey study using typologies of varying precision

LULC type	Low value (2004 $/ac-yr)	High value (2004 $/ac-yr)	Range (2004 $/ac-yr)	% narrower range using more descriptive typology
Beach	**20,969**	**76,416**	**55,447**	
Beach near rental dwellings, receiving recreational use	41,910	77,474	35,564	35.9%
Beach near residential dwellings, receiving recreational use	41,649	76,547	34,898	37.1%
Beach not near dwellings, receiving recreational use	20,704	42,678	21,974	60.4%
Cropland	**6**	**37**	**31**	
Estuary/tidal bay	**18**	**2,854**	**2,836**	
Estuary - large-area, commercial fishery	18	216	198	93.0%
Estuary - small-area, commercial and recreational fishery	236	2,880	2,644	6.8%
Forest	**71**	**5,133**	**5,062**	
Forest – urban proximate	228	5,133	4,905	3.1%
Forest – urban distant	71	1,673	1,602	68.4%
Freshwater wetland	**6,159**	**12,970**	**6,811**	
Open water	**29**	**2,354**	**2,325**	
Open water – large area	182	2,354	2,172	6.6%
Open water – small area	29	741	712	69.4%
Pasture/Grassland	**11**	**17**	**6**	
Riparian zone	**18**	**13,251**	**13,233**	
Riparian – high water pollution levels	30	2,350	2,320	82.5%
Riparian – moderate water pollution levels, urban proximate	4,112	13,251	9,139	30.9%
Riparian – moderate water pollution levels, urban distant	1,811	3,956	2,145	83.8%
Saltwater wetland	**230**	**3,227**	**2,997**	
Saltwater wetland – not near point source	2	954	952	68.2%
Saltwater wetland – near municipal point source	105	1070	965	67.8%
Saltwater wetland – near industrial point source	1,258	17,516	16,258	-442.5%
Urban open space	**1,213**	**4,291**	**3,078**	
Urban or barren	**0**	**0**	**0**	

LULC types in **bold** are from the original LULC typology
LULC types not in bold are from the expanded LULC typology

4. Discussion

Our review of meta-regression studies shows that the authors of these studies have clearly been using contextual variables to describe the socioeconomic and ecological setting of the resources that are being valued, although not consistently. The authors of these meta-analyses frequently acknowledge that limited and inconsistent reporting makes meta-analysis difficult and reduces the number of variables for potential contextual variables that can be included in the regression equations (Woodward and Wui 2001, Brouwer et al. 2003) – a strong argument in favor of use of more systematic contextual variables and LULC typologies in conjunction with a system of metadata for primary ecosystem service valuation studies. Although contextual information is often poorly or inconsistently reported, many studies do contain some of the information necessary to inform a more precise natural capital characterization for ecosystem services value transfer. A more systematic way is now needed to conceptualize and use these contextual variables.

By analyzing typologies used in a past value transfer exercise, we demonstrated that more descriptive cover classes produce less variability in value ranges than the less descriptive categories common to existing LULC systems. More descriptive systems better define the natural capital being valued – its ecological characteristics, landscape settings, scarcity, and the socioeconomic characteristics of and proximity to beneficiaries. New efforts like the USGS ecosystem mapping program demonstrate the promise of high precision, standardized ecosystem mapping to support valuation of ecosystem services (USGS 2008). These maps incorporate diverse biophysical spatial data to systematically map ecosystems at continental scales. When combined with socioeconomic data to map use and demand for ecosystem services, such maps may become even more useful. Better, more uniform definition of the resource being valued is one way to reduce the range of error in value transfer. For example, by identifying and separating wetlands adjacent to a point source polluter from other wetland categories, we improve the theoretical and practical underpinnings of the overall LULC typology. We did, however, find that large value ranges persisted even in our expanded typology, especially for "high value" LULC types where a wide value range may actually exist. In some cases, larger ranges occurred for better defined, higher value resources. In these cases we felt justified using higher value estimates than for more poorly defined, broad cover types (e.g., wetland near point source polluter, versus undefined wetland).

LULC type typically serves as the link between study site and policy site in value transfer. We have endeavored to demonstrate the importance of incorporating more contextually relevant descriptive information in these typologies to facilitate value transfer. We believe such improved definition will reduce transfer error resulting from inconsistent units, though it will not eliminate differences of measurement, methods, and change over time (Rosenberger and Stanley 2006). However, this approach is made more challenging by a general lack of documentation in the literature. Loomis and Rosenberger (2006) call for more consistent and precise reporting of primary valuation studies, and list several important ecological and socioeconomic criteria that should be reported by the authors of primary studies. Consistent reporting of this type of metadata would greatly improve the capacity of both meta-analyses and value transfer to better apply ecosystem service valuation in policy settings. Consistently used, applied, and reported definitions for the resource, its setting, and its user population characteristics is highly desirable in order to improve the theoretical validity of applying value transfer (Banzhaf and Boyd 2005, Loomis and Rosenberger 2006, Spash and Vatn 2006). However, past calls for improved reporting have not been accompanied by any concrete proposals for a specific system of standards for reporting the biophysical and socio-economic contextual variables of the study area. Such a system would allow researchers conducting secondary research and meta-analysis to easily pinpoint the appropriate studies through simple metadata queries across multiple attributes. In doing so, this system could help both to facilitate greater precision in value transfer and to avoid misuse of studies. These metadata could be easily included as online supporting material in electronic journals.

Most past value transfer studies have generally used point transfers rather than function transfers (Loomis 1992). Combining value transfer with the functions identified in ecosystem services meta-analyses has the potential to improve both the theoretical grounding of such exercises and the accuracy of estimates. Improved reporting of well-defined, consistently understood ecological and socioeconomic contextual variables in both primary studies and meta-analyses will improve the likelihood that useful meta-analyses can be developed and applied toward function transfer. To maximize their effectiveness in spatially explicit value transfer (Troy and Wilson 2006), contextual variables should be compatible with GIS data. Using computerized decision support systems that incorporate functions to "intelligently" map, assess, and value ecosystem services across the landscape is another potential application of these techniques at the frontier of ecosystem services research (Villa et al. 2007, Villa et al. in review).

Researchers in ecosystem services should now move beyond general calls for improved reporting in primary studies or better resource definition. What is needed is a flexible, descriptive, well-defined system to catalog, report, and use contextual variables that better define the natural capital being valued in ecosystem services valuation. Ideally, this systematic, precise LULC classification system should be adaptable, improving through use, modifications, and improvements proposed by the diverse collection of scientists, NGOs, and policymakers. These groups will all benefit from a system that helps to better estimate the value of Earth's natural capital and the critical ecosystem services it provides to human societies, from local to global scale.

Acknowledgements

Financial support for this work was provided in part by the Gordon and Betty Moore Foundation grant "Ecosystem services: Dynamics, modeling, and valuation to facilitate conservation." We are grateful to the Project Investigators who provided constructive commentary, including Roel Boumans, Robert Costanza, Serguei Krivov, and Matthew Wilson, as well as to the many graduate research assistants working on the project. We are also greatly indebted to the participants, too numerous to mention, of the summer 2005 workshop at the National Center for Ecological Analysis and Synthesis entitled Understanding, Valuing, and Managing Dynamic Ecosystem

Services under Stress: Synthesizing across the LTER Network, all of whom contributed to the ideas behind this paper.

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CHAPTER 6: FROM ECOSYSTEMS TO PEOPLE: CHARACTERIZING AND MAPPING THE BENEFICIARIES OF ECOSYSTEM SERVICES[1]

Abstract

Ecosystem services research to date has largely focused on the "supply side" – the provision of economic benefits from ecosystems to humans. By comparison, the "demand side," or use of and demand for ecosystem services, has received less attention. This is particularly true for studies that attempt to map and value ecosystem services across the landscape. We argue that the view of ecosystem services popularized by the Millennium Ecosystem Assessment and others (supporting, regulating, provisioning, and cultural services) limits the clear conceptualization of how humans benefit from ecosystem services. Several authors have recently called for such a reconceptualization of ecosystem services. We first review and synthesize these authors' arguments, identifying points of consensus. We find support for an alternate view of ecosystem services as the providers of concrete benefits to specific groups of human beneficiaries. We next explore the linkages between benefits provision and beneficiary use for two ecosystem services – carbon sequestration and storage and aesthetic value, each characterized by different groups of beneficiaries and means of benefit flow from ecosystems to beneficiaries. Finally, we demonstrate how the ARtificial Intelligence for Ecosystem Services (ARIES) system can map ecosystem

[1] Bagstad, K.J., G. Johnson, F. Villa, S. Krivov, and M. Ceroni. In preparation. From ecosystems to people: Characterizing and mapping the beneficiaries of ecosystem services. Target journal: Ecological Economics.

services supply and demand, extending the spatial mapping of ecosystem service provision undertaken by past studies. The resulting beneficiary maps can be combined with provision maps and models to describe how benefits flow from ecosystems to beneficiaries. Such provision, use, and flow maps can greatly advance both the science and policy applications for ecosystem services.

Keywords

Ecosystem services, ecosystem benefits, beneficiaries, demand side, spatial flow, mapping

1. Introduction

Although the ecosystem services literature has proliferated in recent years, the modern concept of ecosystem services can be traced back to at least the early 1970s (SCEP 1970, Mooney and Ehrlich 1997). Since the late 1990s, however, several well-known studies have codified ecosystem services into generally accepted lists or typologies (Costanza et al. 1997, Daily 1997, De Groot et al. 2002, MA 2005). A recent example, the Millennium Ecosystem Assessment, classified ecosystem services into "supporting services," the ecological processes and functions that generate other ecosystem services, "regulating services" that maintain global and local conditions at levels appropriate for human survival, "provisioning services" that offer physical resources directly contributing to human well-being, and "cultural services" that satisfy psychological, emotional, and cultural needs (MA 2005).

The MA classification has proven to be useful for communicating nature's importance in satisfying different domains of human well-being. Yet several authors have recently noted that the MA ecosystem services classification does not lend itself well to economic decision-making (Hein et al. 2006, Boyd and Banzhaf 2007, Wallace 2007, Mäler et al. 2008; hereafter HBBWM). This is because the MA categories do not explicitly link specific benefits and to the human beneficiaries of ecosystem services. Improved definition of these benefits and beneficiaries, combined with their spatial mapping, could aid in ecosystem service valuation, environmental accounting (Boyd and Banzhaf 2007), identification of winners and losers in conservation and development choices (Chan et al. 2007), and in supporting payments for ecosystem services programs (Salzman 2005).

From a spatial perspective, the supply side of ecosystem services has been relatively well-explored. A number of recent studies have used GIS analysis to measure the ecological determinants of value for certain services (Boyd and Wainger 2003, Chan et al. 2006, Naidoo and Ricketts 2006, Beier et al. 2008, Egoh et al. 2008, Grêt-Regamey et al. 2008, Wundscher et al. 2008, Wendland et al. in press). These studies explore how the provision of ecosystem services varies across the landscape. However, far fewer studies have explicitly identified demand side, or human beneficiaries (Hein et al. 2006) or mapped these beneficiaries (Beier et al. 2008)[5]. Yet the need for such mapping is becoming increasingly recognized (Chan et al. 2007,

[5] Though Boyd and Wainger (2003) identify spatial determinants of both supply and demand for ecosystem services, and others have used the MA framework to qualitatively describe beneficiaries (Maass et al. 2005).

Cowling et al. 2007, Naidoo et al. 2008). Supply and demand side mapping are complex, since ecosystem services provision and use often occur across different spatial and temporal scales (Limburg et al. 2002, Hein et al. 2006, Ruhl et al. 2007).

Further, Tallis et al. (2008, pp. 9463) describe the spatial flow problem in ecosystem services. The ecosystem services research community has as yet been unable to move beyond "static maps" to consider cross-scale flows of ecosystem services to different groups of human beneficiaries. Existing attempts to categorization break ecosystem services into coarse categories based on how their benefits spatially flow to beneficiaries, but stop short of providing a quantitative conceptualization (Costanza 2008, Fisher et al. 2009). One way to advance the research and conceptualization of ecosystem services could start from the concepts of the MA (2005) framework, incorporate key elements proposed by HBBWM, and move towards approaches that quantitatively assess spatio-temporal flows of clearly identified benefits from ecosystems towards clearly identified beneficiaries.

Johnson et al. (unpublished) describe how Generalized Source-Sink Models (GSSMs) can characterize the flow of a matter, energy, or information carrier quantity from a source area while determining the sink dynamics resulting from that flow. GSSMs are ideal for modeling ecosystem services, since flows for each service can be based on uniquely defined flow characteristics between regions of provision and use (Table 1). The benefit received by a beneficiary may accrue from receipt of a carrier quantity at the beneficiary's location (as in the receipt of aesthetic views or proximity to open space), or from the absorption of a negative carrier quantity en route to the beneficiary (as in the mitigation of flood waters or landslides).

In order to use GSSMs, we must be able to quantify benefits provided by ecosystem services and the spatial location of their beneficiaries. The ARtificial Intelligence for Ecosystem Services (ARIES) system (Villa et al. unpublished) is a recently developed tool that couples probabilistic models of ecosystem service provision and use with GSSM models to quantitatively assess ecosystem service flows under the following definition:

Ecosystem services are the effects on human well-being of the flow of benefits from an ecosystem to a human endpoint at given extents of space and time.

Table 1: Flow characteristics of selected ecosystem services

MA ecosystem service	Carrier flow	Factors mediating flow	When it stops	Impact on beneficiary, movement after contacting beneficiary
Disturbance regulation: landslide, mudslide, or avalanche regulation	From high to low elevation	Sediment, snow, or ice is held in place, moves downhill, or is deposited, depending on vegetation, topography, and flow characteristics	When it reaches an area with a combination of flatness and/or high enough surface roughness to stop flow	Causes damage to life and/or property. Flow may continue or stop when it's deposited.
Disturbance regulation: flood regulation	Downstream as precipitation falls or snow melts, water travels down the landscape	May be absorbed into the soil or detained on the surface; if neither of these happen it continues downhill	Upon reaching a lake, ocean, or other depression, or if the flow quantity becomes too small to progress to the next downhill cell (i.e., becomes effectively absorbed/detained)	Causes damage to life and/or property if flow is above a certain depth or intensity. Flow continues downhill.
Carbon sequestration & storage	From areas that sequester or store carbon to global beneficiaries	As a well-mixed gas CO_2 and other GHGs enter the atmosphere and move globally	n/a	Users receive benefit of enhanced climate stability, based on relative size of carbon sink
Aesthetic value: views	From scenic places toward locations people inhabit	Follows lines of sight, dependent on topography, aspect, distance, obstructions	When view are blocked or become too degraded or distant	User enjoys sensory benefit. Values are nonrival, so agent keeps moving outward.
Aesthetic value: open space proximity	From open space toward places people inhabit	Follows distance of travel for proximity to open space	When potential users are too far from open space to get there easily (e.g., walking distance)	User enjoys sensory benefit, privacy, and easier access to open space. Values are nonrival, so agent keeps moving outward

In this paper, part of a series of contributions aimed to fully document the ARIES approach, we: 1) review and synthesize the recent contributions of HBBWM in redefining ecosystem services as benefits to human beneficiaries (sections 2 and 3); 2) describe a method for explicitly defining ecosystem services as benefits to people and spatially mapping beneficiaries (section 4); and 3) demonstrate this process by identifying benefits and beneficiaries for two ecosystem services – carbon sequestration and storage and aesthetic value – and mapping their provision, use, and spatial flow (section 5).

2. Recent conceptualizations of ecosystem services

2.1 Millennium Ecosystem Assessment: Supporting, regulating, provisioning, cultural services

As previously discussed, the Millennium Ecosystem Assessment (2005) divided previous lists of ecosystem services (Costanza et al. 1997, Daily 1997, de Groot et al. 2002[6]) into four classes – supporting, regulating, provisioning, and cultural services. A primary goal of the MA was to draw connections between these services and human well-being. Although a useful communication framework, the MA ecosystem service classification lends itself poorly to economic valuation, green accounting, and ecosystem services-based decision making, as described further in sections 1 and 3.

[6] Similar to the MA, de Groot et al. divided their list of ecosystem services into four categories as well – regulation, habitat, production, and information services.

2.2 Hein et al.: New treatment of supporting and regulating services

Through a case study that evaluated wetland ecosystem services in the Netherlands, Hein et al. (2006) assess a set of services, focusing on avoidance of double counting. Double counting can occur if the benefits provided by both an input to an ecosystem service or other economic process and the final good or service are both added to an overall accounting of economic value. As such Hein et al. do not evaluate the MA supporting services, and consider only a limited subset of regulating services. They argue "that regulation services should only be valued if: (1) they have an impact outside the ecosystem to be valued; and/or (2) if they provide a direct benefit to people living in the area[7] (i.e., not through sustaining or improving another service)."

2.3 Boyd and Banzhaf: Final and intermediate services in national accounting

Boyd and Banzhaf (2007) provide perhaps the most thorough critique of the ecosystem services concept, largely in the context of national accounting and development of Green GDP measures. Like HWM, Boyd and Banzhaf consider the separation of "final services" from "intermediate goods and services" to be important. Final ecosystem services can be used in production functions, along with labor and capital, to enable valuation of an ecosystem's contribution to the final market good or service. Intermediate ecosystem services – the ecosystem processes or structure leading to the production of final services – should not be counted, in order to avoid

[7] Although "the area" would differ depending on the scale of the service measured. For example, "the area" would be global in the instance of carbon sequestration and storage, but downstream in the instance of flood regulation. As long as one can make the case that there are direct human beneficiaries somewhere on the globe, we see it as reasonable to include "regulating" services in such an ecosystem services assessment.

double counting. In Boyd and Banzhaf's system, ecosystem services do not include benefits that require contributions from both natural and built capital (e.g., recreation or aesthetic value, both of which depend on both natural and built capital). However, the contribution of natural capital toward the production of recreational or aesthetic value would be counted as a final ecosystem service.

Boyd and Banzhaf consider ecosystem structure – such as a water body, bass population, or riparian forest, to be final ecosystem services (i.e., in contributing to recreational fishing). While their classification of certain elements of ecosystem structure as ecosystem services has raised some objections (Fisher and Turner 2008), Boyd and Banzhaf note that these stock elements of ecosystem structure serve as proxies for the actual service. Final services supply specific human benefits, including positive contributions to human well-being and avoidance of negative consequences that detract from well-being.

2.4 Wallace: Benefits supporting human values

Wallace (2007) also argues that the MA and similar classifications fail to distinguish between "means," processes that generate ecosystem services (e.g., intermediate services, including all supporting and some regulating services) and "ends," the final services that directly benefit humans. Like BBM, Wallace is concerned with the implication of service definitions on ecological and economic management. Wallace gives the example that a natural resources manager does not seek to maximize a mixed basket of intermediate and final services, but to manage certain intermediate ecological processes that will generate certain final ecosystem services that produce tangible benefits for people.

Wallace defines ecosystem services "in terms of the structure and composition of particular ecosystem elements (expressed as assets), and these services are in turn classified according to the specific human values they support." These human values, many of which were also highlighted in the MA, include provision of adequate resources, protection from predators, disease, and parasites, a benign physical and chemical environment, and socio-cultural fulfillment. In Wallace's classification, these four groups of human values are provided in part by certain ecosystem services, which are themselves generated by ecosystem processes. For example, ecosystem processes such as soil retention, nutrient regulation, waste regulation, and climate regulation help provide clean potable water, meeting the human need of adequate resources for survival.

2.5 Mäler et al.: Intermediary, final, public, and private ecosystem services

Like Boyd and Banzhaf, Mäler et al. (2008) evaluate ecosystem services and their classification mainly in the context of green national accounting systems. Mäler et al. reclassify the MA's supporting and regulating services into "intermediary services" and combine provisioning and cultural services as "final services." For accounting purposes, final services contribute directly to human well-being while intermediate services support the production of final services. Accounting for the value of both would constitute double counting.

Mäler et al. also note the importance of classifying services based on rivalness and excludability. These intrinsic characteristics of goods and services strongly affect their management and the institutions needed to promote their sustainable use (Samuelson 1954, Bromley 1991, Ostrom et al. 1999, Farley 2008). Most final services are rival and excludable, making them market goods. This is not the case for most intermediate services, which are nonrival and/or nonexcludable. As such these services are recognized as open access regimes or pure public goods (Daly and Farley 2003).

3. Ecosystem services versus benefits

Overall, HBBWM provide important insights into ecosystem service measurement, mapping, and management (Table 2). These authors seek to classify supporting and many regulating services as intermediate services, or as ecosystem processes that simply support production of final ecosystem services. These final services in turn provide specific benefits to people.

Table 2: Previous conceptualizations of ecosystem services outside the MA framework

<table>
<tr><th>Author</th><th>Abiotic inputs</th><th>Intermediate services/ Ecosystem processes</th><th>Ecosystem structure</th><th>Final services</th><th>Benefits</th><th>Human needs met</th></tr>
<tr><td rowspan="6">Boyd and Banzhaf 2007 (From Table 1)</td><td rowspan="6">-</td><td rowspan="6">Ecosystem processes or structure leading to generation of ecosystem structure or final services</td><td colspan="2">Pollinator populations, soil quality, shade & shelter, water availability; Target fish, crop populations; Target marine populations; Biodiversity</td><td>Harvests: Managed commercial; Subsistence; Unmanaged marine; Pharmaceutical</td><td rowspan="6">-</td></tr>
<tr><td colspan="2">Natural land cover in viewsheds; Wilderness, biodiversity, varied natural land cover; Relevant species populations</td><td>Amenities & fulfillment: Aesthetic; Bequest, spiritual, emotional; Existence benefits</td></tr>
<tr><td colspan="2">Air quality, drinking water quality, land uses or predator populations hostile to disease transmission; Wetlands, forests, natural land cover</td><td>Damage avoidance: Health; Prosperity</td></tr>
<tr><td colspan="2">Surface and groundwater, open land</td><td>Waste assimilation: Avoided disposal cost</td></tr>
<tr><td colspan="2">Aquifer, surface water quality; Aquifer availability</td><td>Drinking water provision: Avoided treatment cost; Avoided pumping cost</td></tr>
<tr><td colspan="2">Relevant species populations; Natural land cover, vistas, surface waters; Surface water, target population, natural land cover; Surface waters, beaches</td><td>Recreation: Birding, Hiking, Angling, Swimming</td></tr>
<tr><td rowspan="3">Wallace 2007 (From Table 3)</td><td rowspan="3">Air, water, land, energy</td><td rowspan="3">Biological regulation, climate regulation, disturbance regimes, gas regulation, management of “beauty,” recreation management, nutrient regulation, pollination,</td><td rowspan="3">“Biotic and abiotic elements”: biodiversity, air, water, land,</td><td>Food, oxygen, water, energy, transportation</td><td>Food, oxygen, water, energy, transportation</td><td>Adequate resources</td></tr>
<tr><td>Benign regimes of temperature, moisture, light, chemicals</td><td>Benign regimes of temperature, moisture, light, chemicals</td><td>Benign physical & chemical environment</td></tr>
<tr><td>Protection from predation, disease, parasites</td><td>Protection from predation, disease, parasites</td><td>Protection from predators, disease, parasites</td></tr>
</table>

		raw materials & medicine production, socio-cultural interactions, soil formation & retention, waste regulation & supply, economic processes	energy	Access to resources for: spiritual & philosophical contentment, benign social group, recreation & leisure, meaningful occupation, aesthetics, capacity for cultural and biological evolution	Access to resources for: spiritual & philosophical contentment, benign social group, recreation & leisure, meaningful occupation, aesthetics, capacity for cultural and biological evolution	Socio-cultural fulfillment
Fisher and Turner 2008 (From Table 1)	Sunlight, rainfall, nutrients, etc.	Soil formation, primary productivity, nutrient cycling	-	Water regulation	Water for irrigation, drinking water, electricity for hydro-power	-
		Photosynthesis, pollination, pest regulation		Primary productivity	Food, timber, nontimber products	

The concept of the *benefit* is a critical insight shared by these authors. Benefits are more specific than broadly defined (MA) ecosystem services. For instance, instead of the confusing and somewhat overlapping "water regulation" and "water supply" ecosystem services listed by the MA, water supply for industry, households, agriculture, recreation, and hydroelectric generation can be accounted for separately. These benefits each have tangible human beneficiaries, which can be spatially mapped. Along with these final ecosystem services and benefits, water regulation and supply also provide an intermediate service – fulfillment of ecosystem water needs, which enables the supply of other final ecosystem services. Many of the intermediate ecosystem services, as defined in the MA, support multiple final services, while final services are often "supported" by multiple intermediate services (Figure 1). Intermediate services may thus interact with other inputs (e.g., built or human capital) in a production function to produce a particular benefit as the output. Benefits are thus a much clearer endpoint and unit of measure for ecosystem services. Further, benefits meet particular categories of human needs – either the provision of physical resources, avoidance of undesirable conditions, or satisfaction of psychological and cultural needs, as identified by Wallace (2007).

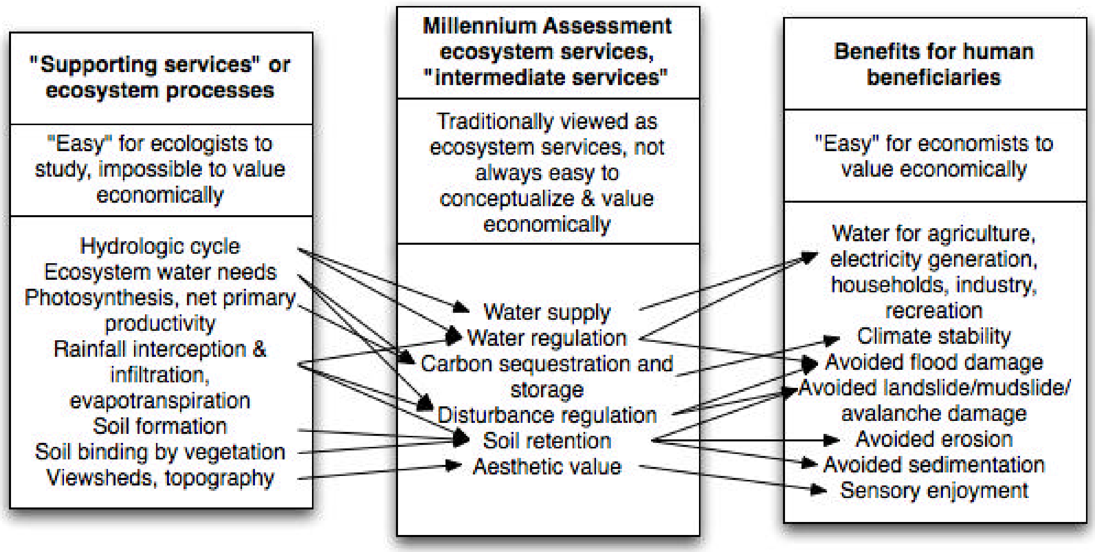


Figure 1: A beneficiaries-based conceptualization of ecosystem services

Fisher and Turner (2008) agree that the distinction between services and benefits is important, and that the benefits concept can advance our understanding of how ecosystem services improve human well-being. Fisher and Turner note that ecosystem processes can directly influence human well-being, and in such cases should be considered ecosystem services (i.e., flood regulation, carbon sequestration and storage). Ecosystem structure can provide a direct benefit (e.g., harvested as timber) or alternatively one or more intermediate services (in providing clean water or flood regulation). Indeed, Farley (2008) argues that the allocation of ecosystem structure between such intermediate and final services is a critical macroallocation issue facing ecological and environmental economics. Fisher and Turner draw clear links between abiotic inputs, intermediate services, final services, and benefits. These benefits could easily be extended to include the human needs met by these benefits, as noted by

Wallace (2007). This conceptualization could also include ecosystem structure (Figure 2).

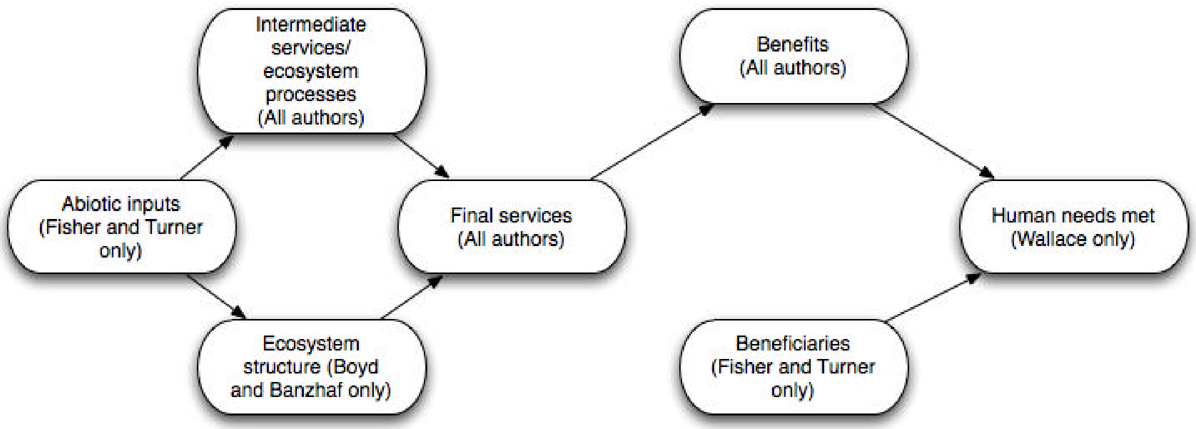


Figure 2: Ecosystem service provision and use: from abiotic inputs to human needs

As Wallace notes, there is a need for clear language in the rapidly growing field of ecosystem services. We thus recognize several important concepts in conceptualizing ecosystem services, on which there is general consensus between HBBWM (Figure 2).

- *Abiotic inputs,* such as sunlight, water, and substrate provide basic resources for ecosystem processes and structure.
- *Ecosystem processes*, often classified as supporting or *intermediate services* are the emergent properties of ecosystems as they process matter and energy. These processes underpin the delivery of final ecosystem services that can be valued by people. Intermediate services are themselves extremely difficult to value,

however. Further, the relationship between these processes and delivery of final services is rarely fully understood by ecologists.

- *Ecosystem structure,* included as Boyd and Banzhaf's *final services,* is the physical configuration of individuals and communities of organisms in nature, including age and spatial distribution and the presence and distribution of abiotic inputs. Ecosystem structure helps generate ecosystem processes; however the extractive use of ecosystem structure also underpins the provision of many ecosystem goods.
- *Final services* provide direct benefits to a group of human beneficiaries, and help satisfy certain human needs. As such they can often be directly valued economically. Final services provided by ecosystems may need to be combined with other capital types in order to produce value. For instance, the recreational value of a water body may depend on the level of access and infrastructure capable of supporting recreation, as well as the recreational preferences of nearby human populations.
- *Beneficiaries* are the specific groups of people who gain from a given ecosystem service. Ecosystem services provide satiation of certain human needs for each group of beneficiaries.
- *Benefits* are the specific gains to human well-being provided by a final service. Benefits can include provision of a basic resource or prevention of an undesirable condition. When valued economically, the opportunity costs of lost benefits are often of interest in a conservation or development decision. For

example, the values of benefits to be lost as part of a land development or resource extraction decision would be conveyed as the opportunity cost of these decisions.

- *Human needs,* or *values* (Wallace 2007) are contributors to human well-being, including satiation of basic needs, protection from undesirable conditions, and maintenance of psychological, social, emotional, and spiritual well-being.

Building on the above consensus points, we conclude that a consistent understanding of ecosystem services is needed to improve communication and decision-making regarding these services. The concepts of benefits and beneficiaries are central, particularly in enabling the mapping of ecosystem service provision, use, and flows.

Within the context of ARtificial Intelligence for Ecosystem Services (ARIES), a web-based decision-support system that uses artificial intelligence for spatial assessment of ecosystem services, such systematic definitions are particularly important (Villa et al. unpublished). Artificial Intelligence (AI) approaches offer promise in improving ecosystem service assessments (Villa et al. 2007), but require expert knowledge to be formalized in ways that AI systems can understand. The following section describes how the ecosystem service concepts developed above can be used to map ecosystem services provision, use, and flows using ARIES.

4. Identifying and mapping provision and beneficiaries: The ARIES approach

4.1 Ontologies for mapping ecosystem service provision and use

ARIES is a decision support system built around probabilistic models of ecosystem service provision and use. Identifying and mapping beneficiaries has been a key step in development of the ARIES system (Villa et al. unpublished). Bayesian networks (Cowell et al. 1999) are used to map the ecological and socioeconomic factors contributing to the provision or use of ecosystem services. These networks enable the use of corresponding GIS data to produce maps of ecosystem service provision and use. Flow models are then used to identify the strengths of ecosystem service flows that provide benefits from ecosystems to people.

As part of the ARIES approach, we use ontologies to define ecosystem services and their provision and use processes. Ontologies are designed to create common, mathematically formalized language for abstract concepts and relationships, promoting consistent, precise, and standardized understanding in a given field (Gruber 1993, Madin et al. 2008). Within ARIES, ontologies also act as a foundation for Bayesian network modeling. They provide a knowledge base for reasoning algorithms to extract models that are then applied to data to quantify how ecosystem services are provided and used (Villa et al. unpublished). As such the ontologies incorporate “ecological production functions” that evaluate the contribution of key influences toward the provision of ecosystem services (Nelson et al. 2009).

In the ARIES context, ontologies specify the following knowledge needed to map ecosystem services provision and use:

1. A core vocabulary for ecosystem services, defining and classifying the general means of provision and use so that specific vocabularies can be built for each service.
2. For each ecosystem service, the breakdown of specific, quantifiable, and spatially mappable benefits that the service produces, the corresponding groups of beneficiaries for each benefit, and the nature of the matter, energy or information carrier that transmits the benefit through space and promotes its transfer to humans (e.g., CO_2 for carbon sequestration and storage, flood water for disturbance regulation, or scenic views for aesthetic value).
3. For each benefit, the set of ecological and socioeconomic attributes that must be observed in order to identify cases of provision and use, so that an appropriately annotated database can be queried to find the data needed for modelling.

Based on the data available, ARIES builds Bayesian network models to describe the likelihood of ecosystem service provision and use, and calibrates them so that a statistical distribution of the final levels of provision and use can be generated for each benefit. Calibration of the model occurs by "training" the Bayesian networks to actual data from (when possible) the region in question, or to known data from regions that are selected by the AI engine from most likely candidates to express similar dynamics as the region of interest. Bayesian models are probabilistic and track the propagation of uncertainties coming from inaccurate or missing data. Thus a full set of input data makes better predictions but is not a prerequisite to running the models (Villa et al. unpublished).

4.2 Mapping ecosystem service flows

Once maps have been generated to show the spatial distribution of provision and use for a given ecosystem service, they are linked by Generalized Source-Sink Models (GSSMs, Johnson et al. unpublished). GSSMs assess the flow of benefits from an ecosystem providing a service to a beneficiary for that service.

To map ecosystem service flows, GSSMs determine "flow districts" by overlaying areas of uniform provision and use, then simulating the trajectory of the matter, energy, or information carrier that transmits each benefit. Each carrier moves according to rules specific to that benefit (e.g., moving downhill, along a line of sight or Euclidian distance, with the flow of water, Table 1). Quantitative data about the strength of benefit flows are computed as the carrier comes in contact with beneficiaries or "sink" landscape features that cause the carrier to be depleted (e.g., sandy soils that cause infiltration of surface flow, visual blight that degrades the quality of views, or the user of a rival service, Table 1). As with all the other models used in ARIES, GSSMs are constructed dynamically by the system using a combination of expert knowledge about ecosystem services and an underlying generic model of the logical and mathematical relationships between concepts that govern how benefits move across the landscape. For each ecosystem service, GSSMs enable the calculation of "provisionsheds" – showing the parts of the landscape from which a given beneficiary's benefits are derived, and "benefitsheds" – showing parts of the landscape where the benefits generated by a particular ecosystem flow toward.

4.3 Flows and economic valuation

Along with providing more realistic views of the biophysical provision of ecosystem services and the benefits they provide to humans, flow maps generated by GSSM models can also facilitate economic valuation. Economic value is the value – monetary or otherwise – of the benefits provided by the marginal unit of the ecosystem service flow in question. The presence or absence of ecological thresholds, where an ecological system abruptly shifts to an entirely different state, has long been realized as important concept in ecology (Holling 1973) and is being increasingly realized as important in economics (Farley 2008, Mäler 2008). In the absence of ecological thresholds, a linear relationship may exist between marginal value and the flow of benefits, simplifying to a classical demand curve. However, if thresholds are present, demand curves may be nonlinear, with value growing exponentially as the demand curve asymptotically approaches the threshold (Farley 2008). At the threshold itself, marginal value becomes total value, as a marginal change leads to the collapse of the ecosystem generating the service. A better understanding of the relative strength of flow can identify regions more likely to provide higher or lower levels of value (Boyd and Wainger 2003). ARIES' value transfer system, under development, combines assessments of the strength of flows and the presence of ecological thresholds to more appropriately apply economic values from other regions when local economic value estimates are unavailable.

5. Mapping beneficiaries: A case study

5.1 Approach

Following the connections between MA ecosystem services and beneficiaries described in section 3, we identify the specific beneficiaries and the benefits for two ecosystem services (Table 3). We also identify the spatial data needed to map the location of these beneficiaries. In order to enable modeling in ARIES, all benefits must meet five requirements. Specifically, benefits must be: 1) quantifiable, 2) directly valuable to humans, 3) provided wholly or in part by one or more clearly identified natural entities or processes, 4) used by one or more clearly identified human consumers, and 5) provided through movement of a clearly identified material, energetic or informational carrier. Once the full chain of provision and use is clearly described, there is no need to distinguish “intermediate vs. final services,” “supporting services,” or to worry about double counting in valuation, because the base for valuation is the quantifiable flow of benefits rather than the ecological processes that bring benefits into existence.

Table 3: Mapping beneficiaries for carbon sequestration and storage and aesthetic value

"Traditional" ecosystem service	**Beneficiaries**	**Specific benefit gained**	**GIS data needed**
Carbon sequestration and storage/climate stability	**Groups vulnerable to the effects of climate change**		
	Arid region populations sensitive to changes in precipitation	Reliable sources of water for agriculture, industry, domestic use, electricity generation, recreation	Rainfall, population density, hydroelectric dam locations and size, agricultural land, recreation sites
	Farmers sensitive to changing temperature and climate	Reliable rainfall, surface, and groundwater resources for agriculture	Agricultural land, rainfall, potential evapotranspiration
	Populations in risk zones for forest fires	Avoided property loss due to intense forest fires	Population density, forested areas, fire frequency
	Coastal populations	Avoided loss of property due to inundation from rising sea level	Coastal property, elevation
	Coastal populations	Avoidance of tropical storms of increasing intensity	Elevation, population density, coral reefs, tropical storm tracks
	Mountain zone populations	Avoided landslides due to intense storms	Population density, unstable/steep soils, elevation
	Populations living in permafrost zones	Avoided infrastructure loss/damage from permafrost melt	Permafrost boundary, population density, public infrastructure or private structure locations
	Populations at high latitudes	Avoided exposure to excess UV due to delayed ozone recovery	Ozone hole extent, population density
	Snowmelt dependent populations	Reliable sources of water for agriculture, industry, domestic use, electricity generation, recreation	Snowpack/mountain glacier locations, watershed boundaries, downstream cities (population density), hydroelectric dam locations and size, agricultural land, recreation sites
	Residents of megacities	Avoidance of exacerbated air quality and human health problems – heat waves, ground level ozone, allergen production	Population density, urban extents, regions with current air quality problems
	Poor subsistence populations	The poor, who make their livelihood directly from subsistence living (for food, water, fuel, fibers, etc.), are likely to be	Population density, per capita income, urban extents

		hardest hit by climate change	
	Future generations	The costs of climate change are largely pushed onto future generations	Population growth rates
	All people globally enjoying benefit of climate stability		
	All people, everywhere	All humans are affected to some degree by climate change	Population density
	Groups using atmospheric waste absorption capacity for greenhouse gases		
	Greenhouse gas emitters	Emitters of greenhouse gases, especially those emitting at high levels, disproportionately rely on carbon sequestration and storage to offset their emissions	Population density, greenhouse gas emissions
Aesthetic value	**Groups enjoying scenic views**		
	Residents with scenic views from their house or property	Sensory enjoyment from scenic views from private property	House locations or population density, elevation, locations of scenic features (e.g., mountains, water bodies)
	Park visitors	Sensory enjoyment from scenic views in parks	Scenic viewpoints from parks, visitation rates, surveys on travel distance
	Travelers along scenic routes	Sensory enjoyment from scenic views along roads, passenger rail, ferries	Scenic views from roads, rail, or ferries, traffic estimates, travel distances
	Groups enjoying proximity to open space		
	Homeowners near open space	Sensory enjoyment from adjacency or proximity to open space, privacy, easier access to recreational amenities	House locations or population density, open space type (wetland, forest, grassland, park, beach), protected status, access points

After identifying beneficiaries (Table 3), we next designed ontologies and Bayesian network models to enable spatial mapping of each group of beneficiaries (Figure 3), as exemplified by the traditionally conceptualized ecosystem services of carbon sequestration and storage and aesthetic value. Similar ontologies to enable mapping of beneficiaries of all other ecosystem services are under development as part of the ARIES project. Corresponding ontologies and Bayesian network models have also been developed to enable modeling and mapping of ecosystem service provision.

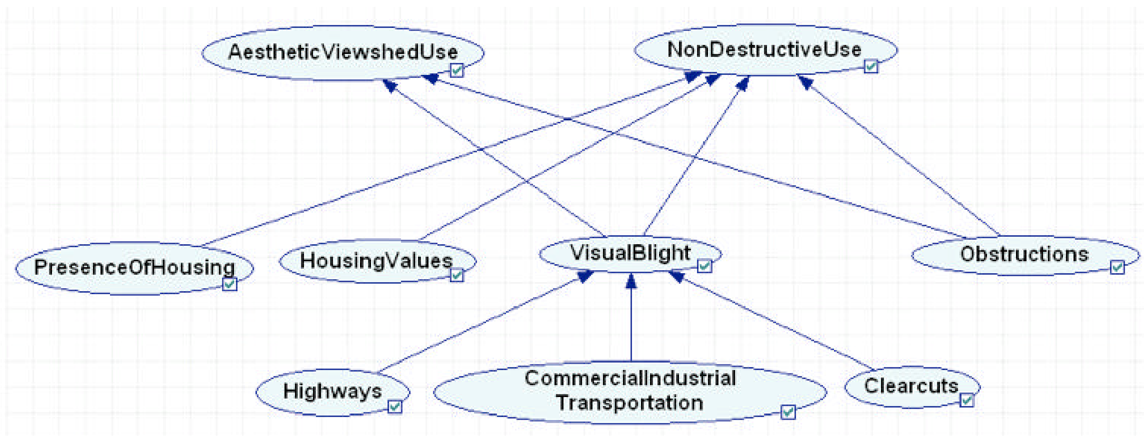


Figure 3: Beneficiary ontology, describing the benefits of aesthetic views, and the spatial data needed to map these beneficiaries

5.2 Ecosystem services and their beneficiaries

Carbon sequestration and storage by ecosystems is increasingly important as society responds to the threat of climate change (Portela et al. 2008). Carbon sequestration and storage help provide a more stable global climate by taking up greenhouse gases and keeping them out of the atmosphere. Specific beneficiaries of a stable climate can be identified, particularly in regions most vulnerable to climate change. We can thus map vulnerable populations that depend on a stable climate as the beneficiaries of carbon sequestration and storage. Regions and human populations that are most vulnerable to climate change are identified in the ecosystem services and climate change literature (MA 2005, Schröter et al. 2005, Stern 2006, Parry et al. 2007). Vulnerable groups include coastal populations at risk of sea level rise and more intense storms, populations dependent on glaciers and snowpack for water supplies, populations in arid regions at risk of drought, and populations dependent on infrastructure built on permafrost, among others. A variety of spatial data layers enable the mapping of these groups vulnerable to climate change.

Alternatively, all people can be recognized as the beneficiaries of a stable climate, since flows of people, goods, and services affect all parts of an interconnected global economy. Assigning all people a right to climate stability or to ecosystem carbon sequestration and storage is also a democratic way to assign value.

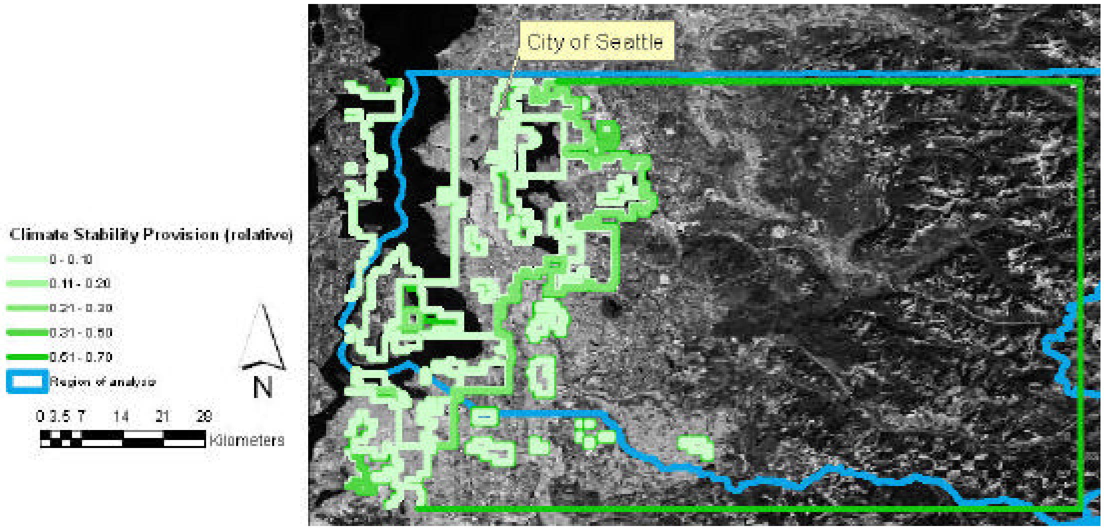
A third way to conceptualize the beneficiaries of carbon sequestration and storage is to examine per capita greenhouse gas emissions, which can then be compared to per capita carbon sequestration in a region to evaluate regional carbon budgets. Since emitters of greenhouse gases benefit from the waste absorption capacity of the biosphere, carbon sequestration and storage can be divided among emitters. Existing and proposed systems to cap and assign property rights to greenhouse gas emissions essentially use this framework.

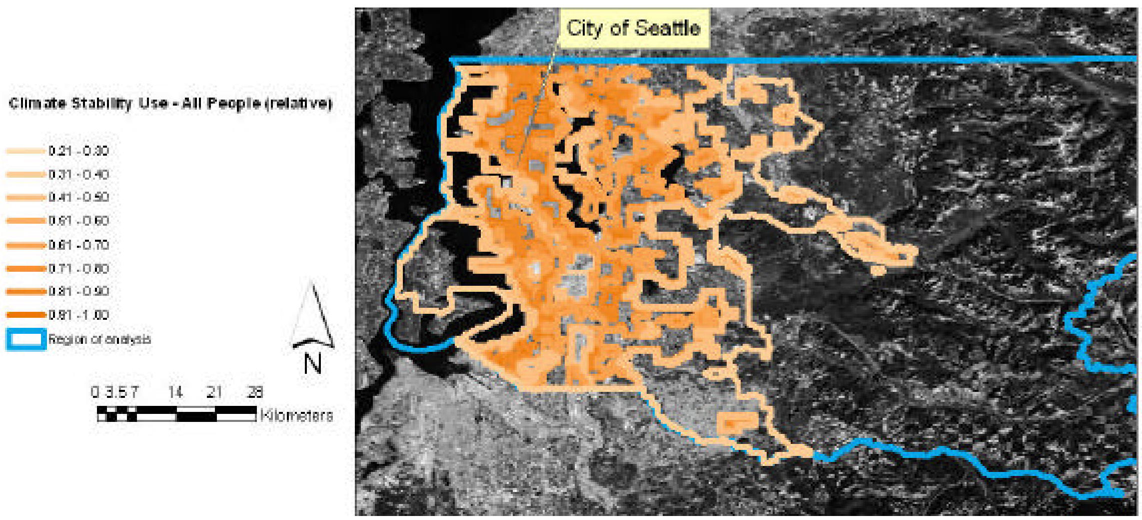
Aesthetic values are typically defined as the value derived from views (Bourassa et al. 2004) or proximity to open space (McConnell and Walls 2005). These values often accrue to housing or property values and can thus be measured using hedonic pricing. Homeowners are typically the beneficiaries of aesthetically valuable property, as they gain some combination of sensory enjoyment from views of nearby open space, proximity to open space for recreation, and privacy. Scenic views are also typically valued by homeowners, but may also be valued by parkgoers or drivers along routes with scenic views (Hallo and Manning in press). Spatial mapping of homeowners is relatively straightforward, as housing units, open space, and viewsheds have all been frequently analyzed in hedonic studies of amenity values. However, the flow patterns of open space proximity and viewsheds differ, with the value of open space declining with distance from the open space resource, while viewsheds depend on a line of sight to a visually significant object, such as a mountain or water body.

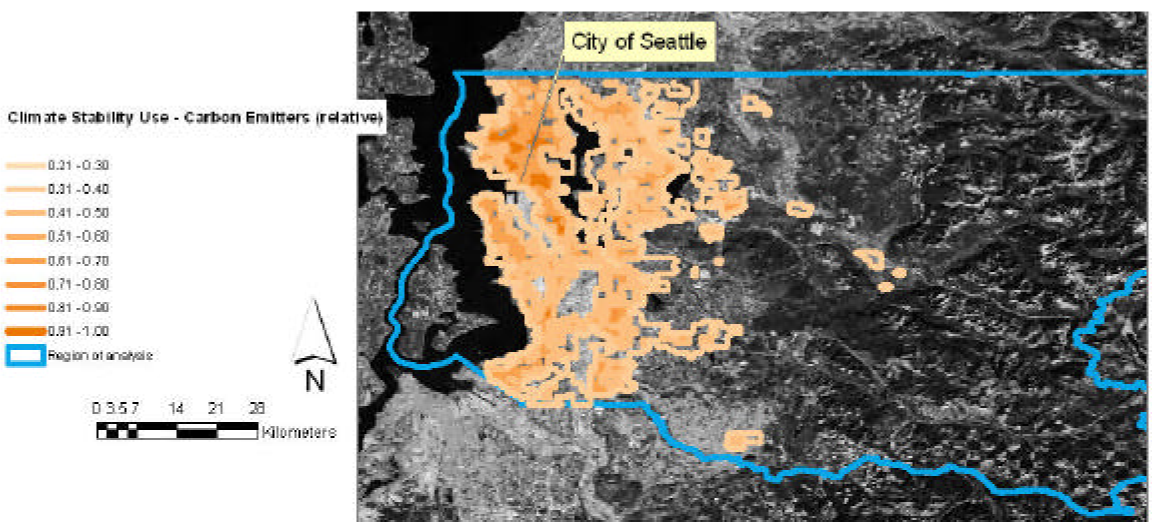
5.3 Mapping beneficiaries

To fully document ecosystem service provision, use, and benefit flows, we first map the regions containing ecosystems that provide a given benefit, and those that contain people that depend on that benefit. Beneficiaries are mapped using data layers shown in Figure 3 and Table 3. Spatial data are incorporated into Bayesian networks encoding ecosystem service supply and demand relationships to produce maps, as described in section 4. We map provision and use of ecosystem services for a part of Puget Sound, Washington State, an ARIES case study region. For each service, four maps can completely describe its provision, use, and flow: 1) a map showing the ecosystems that provide the service, 2) a map showing the location of human beneficiaries of the service, 3) a "provisionshed" map, showing the parts of the landscape from which a given beneficiary's benefits are derived, and 4) a "benefitshed" map showing parts of the landscape where the benefits generated by a particular ecosystem flow toward (Johnson et al. unpublished).

Climate stability is provided through carbon sequestration and storage, and is greatest in the more forested eastern portion of King County, Washington (Figure 4). The use of climate stability by all people and greenhouse gas emitters, by contrast, is located primarily in the more populous western part of King County. Residents of fire-prone regions, who may see the frequency of forest fires increase due to climate change, are one of several vulnerable groups that can be mapped. Other vulnerable groups that could be similarly mapped include coastal populations, farmers, and snowmelt dependent populations.


City of Seattle
Climate Stability Provision (relative)
0 - 0.10
0.11 - 0.20
0.21 - 0.30
0.31 - 0.50
0.51 - 0.70
Region of analysis
N
0 3.5 7 14 21 28
Kilometers


City of Seattle
Climate Stability Use - All People (relative)
0.21 - 0.30
0.31 - 0.40
0.41 - 0.50
0.51 - 0.60
0.61 - 0.70
0.71 - 0.80
0.81 - 0.90
0.91 - 1.00
Region of analysis
N
0 3.5 7 14 21 28
Kilometers


City of Seattle
Climate Stability Use - Carbon Emitters (relative)
0.21 - 0.30
0.31 - 0.40
0.41 - 0.50
0.51 - 0.60
0.61 - 0.70
0.71 - 0.80
0.81 - 0.90
0.91 - 1.00
Region of analysis
N
0 3.5 7 14 21 28
Kilometers

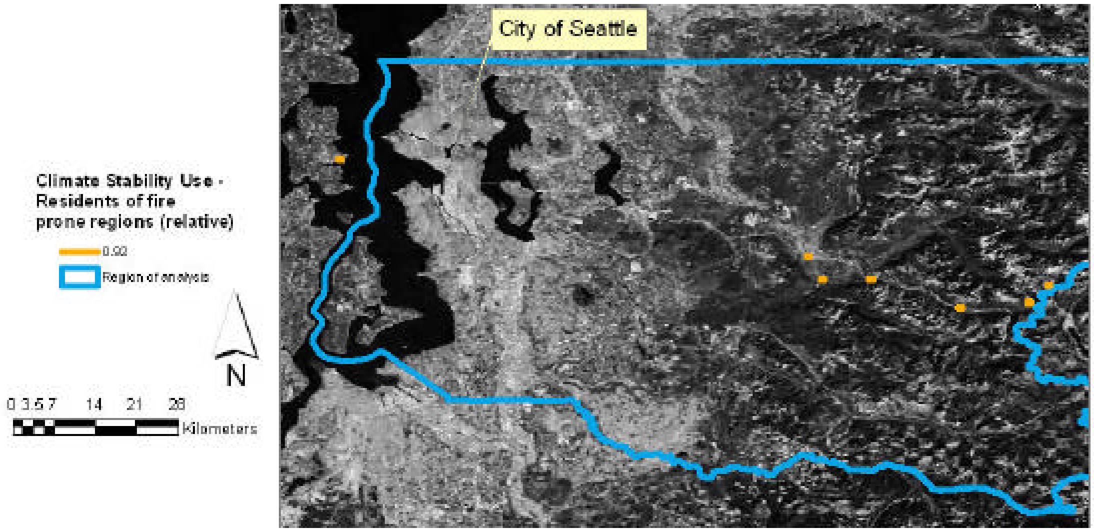
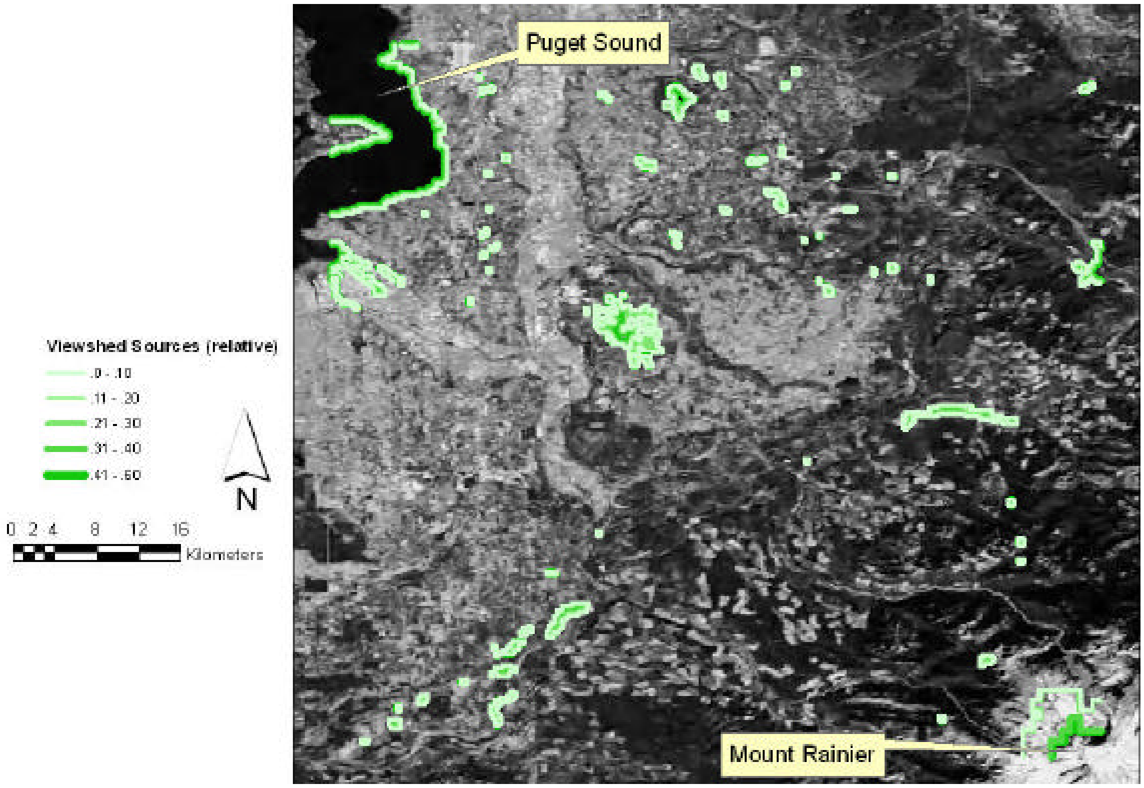
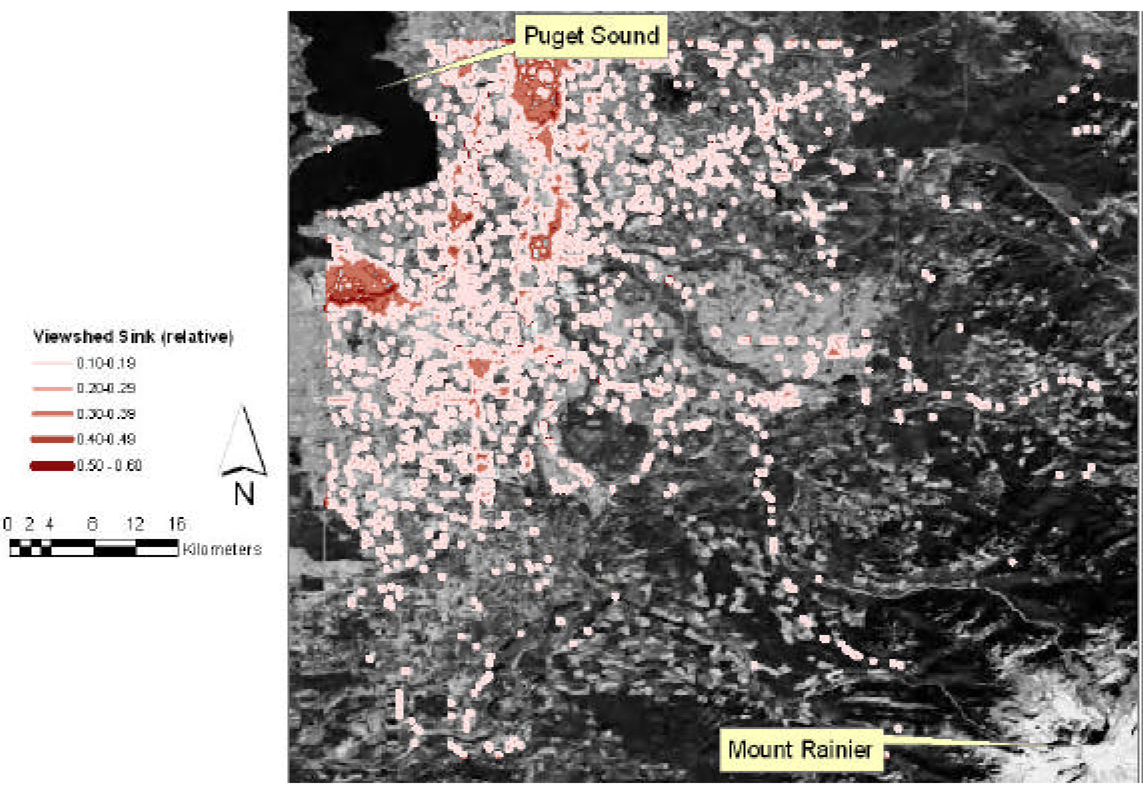


Figure 4: Provision and use of climate stability in part of Puget Sound, WA

Sources of aesthetically valuable views include large mountains like Mount Rainier and water bodies such as Puget Sound or inland lakes (Figure 5). Users of aesthetic views are found in residential areas, such as the City of Kent, Washington. As a view travels from source to user, it may be physically blocked by buildings, trees, or landforms, or its quality may be depleted by air pollution or visual blight, reducing the view quality. Sources of visual blight can also be mapped, including highways, forest clearcuts, and commercial, industrial, or transportation land uses. Higher concentrations of residential users or visual blight lead to higher levels of ecosystem service use or sinks, respectively.


Puget Sound
Mount Rainier
Viewshed Sources (relative)
.0 - .10
.11 - .20
.21 - .30
.31 - .40
.41 - .50
N
0 2 4 8 12 16
Kilometers


Puget Sound
Mount Rainier
Viewshed Sink (relative)
0.10-0.19
0.20-0.29
0.30-0.39
0.40-0.49
0.50 - 0.60
N
0 2 4 8 12 16
Kilometers

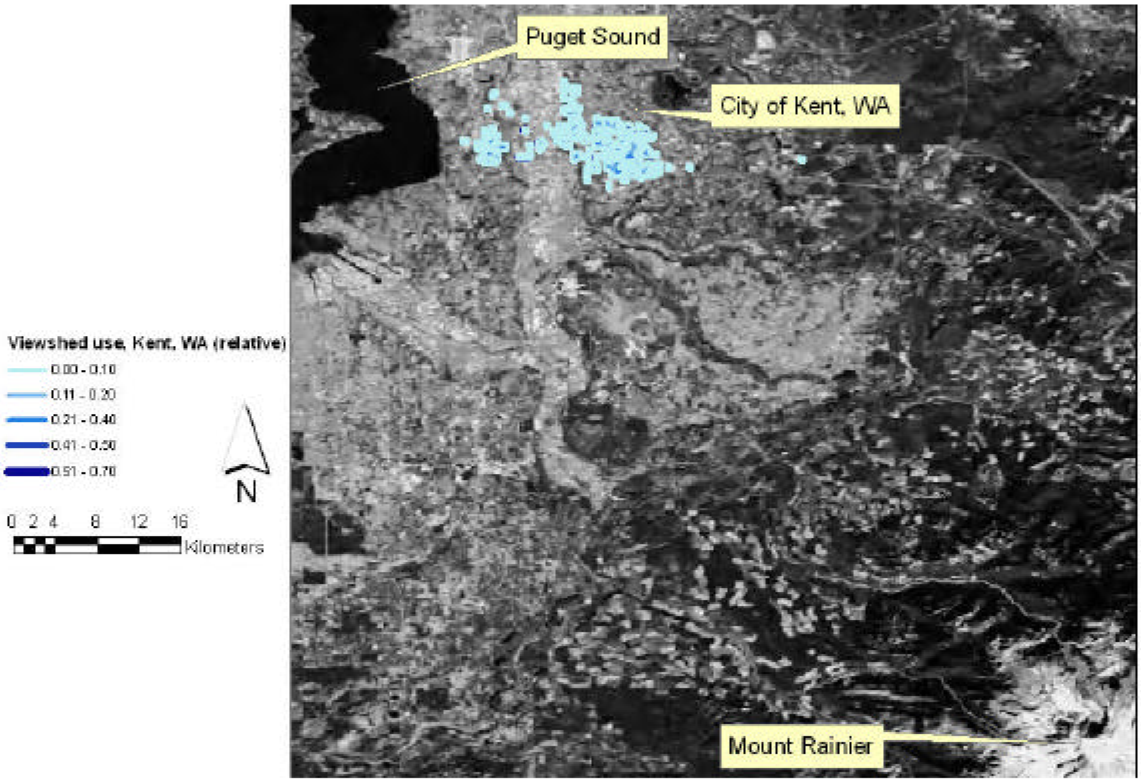


Figure 5: Provision, sinks, and use of aesthetic views in part of Puget Sound, WA

6. Future work and conclusions

We have to date mapped only the sources and beneficiaries of carbon sequestration and storage and aesthetic views. However, upcoming work will enable the mapping of a number of other ecosystem services, including aesthetic proximity value, flood regulation, soil retention, and the provision of salmon as a source of recreation, food, and cultural value. Analysis of multiple ecosystem services enables us to overlay services, identifying areas that provide bundles of multiple services and to compare tradeoffs between services (Chan et al. 2006, Nelson et al. 2009). Future work will also enable mapping of the spatial flow from ecosystems to people, enabling us to demonstrate “provisionsheds” and “benefitsheds” for each service, based on the unique matter, energy, or information carrier for each service.

Understanding the flow pattern of benefits from ecosystems to people is a problem that has eluded past work in ecosystem services (Ruhl et al. 2007, Tallis et al. 2008). For many authors, the flow problem has been expressed as a “spatial mismatch” between ecosystem service provision and use (Limburg et al. 2002, Hein et al. 2006, Costanza 2008, Fisher et al. 2009). By explicitly demonstrating spatial links from ecosystems to people and the strength of the flow of ecosystem services, we can better demonstrate how specific beneficiary groups gain value from ecosystem services. Particularly in the developed world, the beneficiaries of ecosystem services are often unaware of their dependence on ecosystems (Costanza 2008). Mapping of the beneficiaries of ecosystem services and the spatial flows of services are important steps in raising awareness of the value of ecosystem services. This can lead to both fuller appreciation of value by the groups that benefit most from nature’s services, and a better body of knowledge to enable sound decision making by society.

Acknowledgements

Funding for this paper was provided by the National Science Foundation. We thank the ARIES partner organizations, Conservation International and Earth Economics and their stakeholders for providing spatial data and an improved understanding of regional ecosystem services dynamics. Josh Farley led work on how to incorporate economic valuation of ecosystem services in ARIES. Ted Auch provided constructive reviews of ontologies for carbon sequestration and storage.

Austin Troy provided valuable review comments on an earlier version of this manuscript.

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