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Analysis

A Spatially Accurate Method for Evaluating Distributional Effects of Ecosystem Services

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ABSTRACT

The value of most ecosystem services invariably slips through national accounts. Even when these values are estimated, they are allocated without any particular spatial referencing. Little is known about the spatial and distributional effects arising from changes in ecosystem service provision. This paper estimates spatial equity in ecosystem services provision using a dedicated data disaggregation algorithm that allocates 'synthetic' socio-economic attributes to households and with accurate geo-referencing. A GIS-based automated procedure is operationalized for three different ecosystems in Israel. A nonlinear function relates household location to each ecosystem: beaches, urban parks and national parks. Benefit measures are derived by modeling household consumer surplus as a function of socio-economic attributes and distance from the ecosystem. These aggregate measures are spatially disaggregated to households. Results show that restraining access to beaches causes a greater reduction in welfare than restraining access to a park. Progressively, high income households lose relatively more in welfare terms than in low income households from such action. This outcome is reversed when distributional outcomes are measured in terms of housing price classes. Policy implications of these findings relate to implications for housing policies that attempt to use new development to generate social heterogeneity in locations proximate to ecosystem services.

1. Introduction

Ecosystems provide services to households located in their vicinity. Some of these services are not mediated through the market and thus their value is absent in national accounting. For example, if fees are not charged for the use of national parks, the cultural and recreational services they provide are missing from national accounts even though they contribute to the welfare of households. Economists have developed a variety of methods for estimating the value of ecosystem services where market prices are not perfect or do not exist but in general these values are allocated to the different ecosystems without any particular distributional referencing (Costanza et al., 1997). Very little is known about how changes in ecosystem service provision are distributed across population groups, for example, do high income households receive more services than low income households.

The evaluation of ecosystem services is invariably concerned with generating average values for different services and attaching real and shadow prices to a generally unpriced and heterogeneous good. For example, the main concern of the Millennium Ecosystem Assessments (Millennium Ecosystem Assessment, MEA, 2003) and other country level assessments (Bateman et al., 2011; Patterson and Cole, 2013) is to show the degradation of ecosystem services. It has become increasingly clear that global human population and consumption patterns are well above what can be supported without impairing vital life-support systems (Ehrlich et al., 2012). Thus, there is a need to develop mechanisms for integrating the consumption of ecosystem services into land use and resource decisions (Nelson et al., 2009). Because many ecosystems do not have a market value, they are typically undervalued[1] when policies and decisions are formulated and recognized only upon their loss (Daily et al., 2000). The evaluation of ecosystem services provides an economic measure which can be compared to private goods and used in assessment of global change.

Invariably, these assessments are undertaken in aggregate without concern for the issue of who benefits and who loses in the wake of change in ecosystem services provision. This paper contends that considering aggregate change is insufficient and that distributive effects

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[1] The context for this under-valuation is not explored here. The market may blind-side ecosystem services for reasons other than accounting inadequacies. For example, under-valuation may be a case of lack of 'voice'. Those who value ecosystem services may have little political influence or state agencies charged with their protection may have limited funding.

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across population groups need to be addressed. While the general notions of ecosystem services equity and environmental justice do occupy the literature, this attention is commonly focused on theoretical discussions of how to define distributive justice of ecosystem services (Matulis, 2014), macro-level analyses relating to poor and rich nations (Schomers and Matzdorf, 2013) or case studies, for example analyzing poor populations exposed to environmental degradation (Brulle and Pellow, 2006).

This paper deals with the empirical distribution of ecosystem services at the sub-national and local levels utilizing micro (household) data for Israel. It contributes to the methodology of ecosystem service assessment by introducing an approach for accurately assessing the spatial distribution of ecosystem service benefits and evaluating the welfare and distributional effects of changes in accessibility to ecosystem services. To create the high-resolution spatial microdata necessary for such an exercise, recent advances in data disaggregation and the generation of synthetic spatial microdata are exploited. The paper utilizes an allocation algorithm that downscales census tract data into households on a national basis. This allows for the automated calculation of the effects of change in ecosystem services provision at high levels of spatial resolution.

The paper subsequently estimates household benefits arising from the value of recreational visits at the microscale from three types of sites using a simulated consumer surplus[2]. These relate to beaches, national parks and urban parks which each belong to different ecosystems as defined in the Israeli National Ecosystems Assessment (IESA, 2014): marine ecosystem, Mediterranean ecosystem and urban ecosystem, respectively. We identify welfare change that can be linked to distance from the sites and the socio-economic attributes of the households consuming ecosystem services. The disaggregated economic value is embodied in the consumer surplus derived by different population groups. This surplus can be recombined into various welfare measures that show distributional impacts of changes in ecosystem services to different population groups at various spatial scales. The paper thus makes two contributions: methodological and empirical. In terms of method, we present a reproducible approach for the accurate spatial identification and estimation of ecosystem services benefits. The empirical contribution lies in the estimates of distributional and welfare impacts of these benefits under two different policy scenarios.

2. Literature Review

There is a growing theoretical discourse concerning ecosystem services and distributive justice. Sievers-Glotzbach (2013) offers a theoretical framework to consider the distribution of access rights to ecosystem services. She shows that the Rawlsian "Theory of Justice" (Rawls, 2009) can be extended to contain the justice issues of ecosystem services. Accordingly, the argument is that access rights to vital ecosystem services need to afford the greatest benefits to the least advantaged members of the present and actual future generations. Jax et al. (2013) contend that the distribution of benefits and costs associated with the provision of ecosystem services should be calculated across both spatial and temporal scales. Farley (2012) claims that in the case of ecosystem services that cannot be privately owned the principle of equal say for all in allocation decisions concerning ecosystem services should hold. These are all theoretical discussions of what is considered justice in the framework of ecosystem services.

The empirical literature, concerning the evaluation of ecosystems services, adopts two approaches to welfare and distributional issues of ecosystem provision. The first focuses on using payments for ecosystem services (PES) for poverty alleviation (Gauvin et al., 2010; Corbera et al., 2007; Paavola and Lowe, 2005). These payments go to the inhabitants of rural areas in return for supplying ecosystem services by refraining from intensive farming and adopting habitat-protective techniques. Since most of the global poor live in rural areas PES has an equity effect. The poor are paid for reducing environmental damages. The second approach deals with environmental inequalities. Some studies look at environmental disamenities showing how social and economic dynamics result in the poor being more exposed to environmental hazards than the rich (Ringquist, 2005). Others have looked at access to environmental amenities such as parks and open spaces and investigated their equitable distribution (Mitchell and Popham, 2008; Boone et al., 2009). There is a recent literature on environmental (in) justice with a wealth of case studies and some quantified facts in the wake of the work pioneered by Martínez-Alier (2002) and Hornborg and Martinez-Alier, (2016) (e.g., the results of the EJOLT project in Hornborg). Most of this genre analyzes the relationship between income and social attributes to accessibility to environmental disamenities or amenities. Very little attention has been paid to analyzing welfare effects of ecosystem services at a high level of spatial resolution capable of identifying local equity/social welfare outcomes.

Researchers have developed a variety of methods for estimating the value of ecosystem services when market prices are not determined in a perfectly competitive market (i.e. when there are subsidies or taxes) or where market prices do not exist. They include adjusted market prices, production function methods, damage cost avoided, averting behavior, revealed preference methods and stated preference methods (Bateman et al., 2011 Ch. 22 in NEA_UK). In general, these values are allocated without any particular spatial referencing and the main effort is directed towards extracting a value for a hard-to-measure service. The result is a plethora of case-study type investigations that deal with the evaluation of a given service in a particular place (Crossman et al., 2013). Similar research has also been conducted in Israel with a focus on idiosyncratic pricing of site contamination (Shelem et al., 2011) or agricultural landscapes (Fleischer and Tsur, 2009). Nevertheless, Bateman et al. (2013) develop a methodology for spatially sensitive and ecosystem-specific prediction of outdoor recreation visits and their value. Their major objective is the prediction of area-specific recreational value under different scenarios. They combine a trip generating function model and meta-analysis of per visit values to estimate the number of visits and the value of each visit for each 1 km grid square in their study area. Unlike Bateman et al. (2013), our units of analysis are households rather than grids. Furthermore, our focus of concern is the recipients of ecosystem services rather than the service-generating sites. We attempt to predict how these services are distributed among households under different scenarios.

Generally, estimating welfare and distributional effects across different communities is not possible as the data generally do not allow for spatial differentiation. For example, while the conversion of farmland to forests may generate general population-wide services such as carbon storage and land reclamation, it could be that the resultant recreation values might be much more selectively distributed across the relevant nearby urban population. The value of the ecosystem service in this case will be appropriated by a small sub-group of the population with the ability to benefit (the young, the mobile etc.) whereas the welfare of the overall population may not change or may even decrease, resulting in negative distributional outcomes.

3. Method

In order to evaluate the distribution of recreational services that households in Israel elicit from three types of ecosystems, we adopt a four-stage method (Fig. 1). The first stage involves an allocation algorithm that disaggregates census tract data into households and spatially allocates them into dwelling units. The allocation procedure uses the 'synthetic reconstruction' approach (see Hermes and Poulsen, 2012) for artificially generating data and iterative proportional fitting (IPF) for

[2] Consumer surplus is defined as the difference between the willingness to pay for goods and services and the amount actually paid. It is used a measure of welfare change in environmental economics (see Freeman, 1992, p. 48).

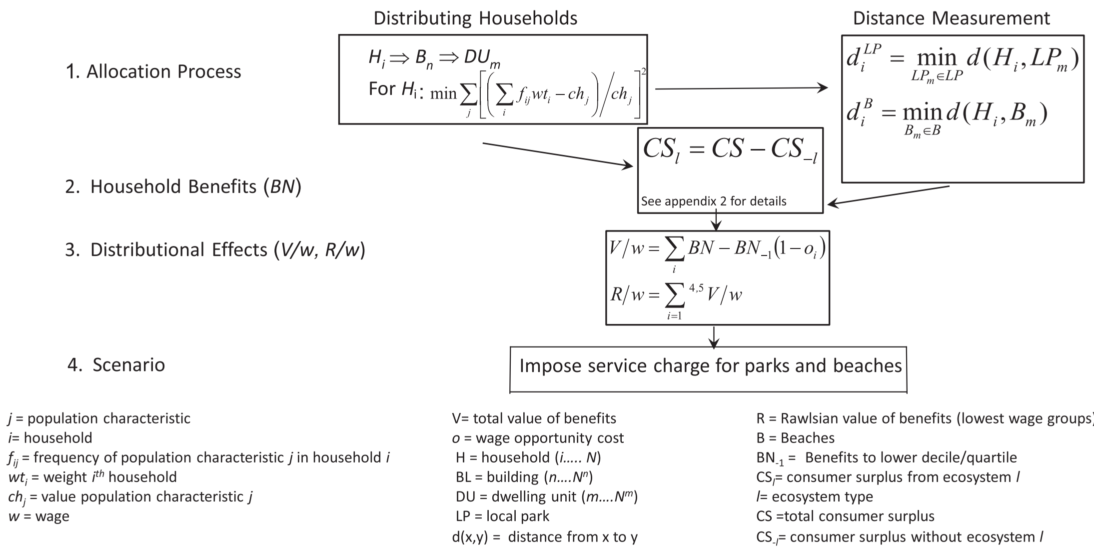

Fig. 1. Ecosystem service evaluation method.

sequentially adjusting the synthetic data so that it corresponds to the known marginal distribution of the population. Given our need to generate synthetic data, this approach is preferable to the alternative of ‘reweighting’ whereby individuals or households from survey data are assigned weights that flag their representativeness in some known aggregate distribution. In our case, the task at hand calls for populating spatial units (buildings) with synthetic populations. This can only be done by selecting households and individuals from random samples such that the joint distribution of the critical attributes of interest in the synthetic population, matches marginal distributions from some aggregate source such as a census (Beckman et al., 1996). IPF is used as the adjustment procedure for estimating joint distributions from a set of control variables. In contrast to standard reweighting, this approach ensures that synthetic households match iteratively determined joint distributions. However, it should be noted that the IPF system can result in a population of households whose distribution matches the marginal distribution and individuals whose distribution does not.

Data analysis and processing procedures are written in Python and SQL and are fully automated. The mechanics of this system are formally described elsewhere (Felsenstein et al., 2016) so we suffice with a short description here.

We create disaggregated discrete data in which each household and individual in a given census tract is represented as a separate entity and allocate socio-economic attributes to these entities. The first allocated attribute relates to age. Individuals in each household are assigned an age category that is iteratively adjusted to represent the age distribution of households in each census tract (CT). The code ensures no anomalies arise and thus no households are comprised entirely of children and each household comprises at least one adult. Gender is another key allocation variable. In contrast to the homogeneity in the adult age distribution allocation, the gender allocation procedure aims at producing gender heterogeneity and ensuring that the synthetic distribution matches the marginal known distribution. Variables relating to workforce participation, occupation, industry of employment, disabilities, education and car ownership are then assigned to households in very similar manner. In most cases, this is according to the age and gender marginal distributions. In contrast, earnings (remuneration from work) are treated differently. They are distributed to households based on a Mincer-type earnings regression that relates to the marginal contributions of age, education, gender, occupation and industry of employment. Finally, we allocate households to buildings (and dwelling units) on the basis of a simple Alonso-style mechanism where each household strives to minimize the difference between its given bid-rent equilibrium and its locational choice implications in terms of price and land-size combinations (see Appendix 1).

Once households are allocated to buildings, the next calculation involves measuring their accessibility to the different type of recreational sites that represent different ecosystems (‘distance measurement’ in Fig. 1). We use a GIS based shortest path algorithm on a national road network that takes into consideration carrying capacity, direction and average driving speeds to measure accessibility to the sites. For each of the 2.3 million national households, the shortest distance to the nearest representative ecosystem site is computed for each ecosystem type. The sum of these distances then enters the household benefit calculation as a travel cost component in the next stage of the analysis. All these calculations are fully automated using an SQL spatial database.

Household benefits (*BN*) from recreational visits to each type of ecosystem site are estimated in the second stage (Fig. 1). We estimate a non-linear function relating household location to each of the three sites. The welfare elicited by each household from these ecosystems is estimated based on Fleischer and Tsur (2003) who use travel cost. An aggregate measure of the recreational value of the three main types of ecosystems in Israel is developed. This relates to beaches, urban parks and national parks. The procedure assumes that households are indifferent between different sites of the same type. This means that households derive utility from the recreational activity in each type of site regardless of the site's specific attributes, except for distance. The estimated consumer surplus is derived from the demand function and thus is contingent on the socio-economic attributes of the household and on distance from the sites. We use parameters from Fleischer and Tsur (2003) to generate the number of site visits attributable to each household and the consumers' surplus from visits to each type of site. The formal representation of consumer surplus from each type of ecosystem appears in Appendix 2.

The spatial distribution of the simulated number of visits to different ecosystem services is presented in Fig. 2.These graphics represent the baseline conditions against which scenarios of change in ecosystem service provision are compared. The highest number of visits are to beaches (panel a) naturally generated by households located near the Mediterranean shore, around the Sea of Galilee or close to the Red Sea. Urban parks are the next most frequented sites due to their proximity to heavily populated areas. National parks are the least visited locations with a more uniform distribution of visits.

The third stage estimates group welfare effects (*V*) by summing the change in utility across earning groups or any other category that

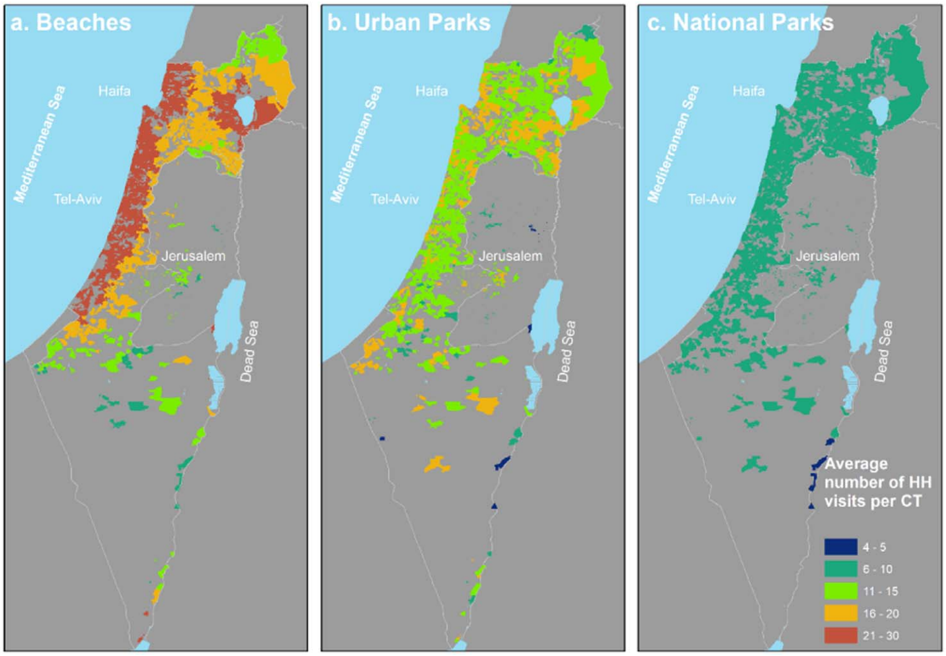

Fig. 2. Average number of household visits to sites representing different ecosystem services, by census tract.

captures welfare gains/loss (Fig. 1). This means re-aggregating the disaggregated data back into manageable units for presentation. These aggregates can be spatial (regions, cities neighborhoods), income or earning classes by deciles or quantiles, housing classes by house price categories and so on. This is done for ecosystem services, jointly or severally. Welfare changes need to be adjusted by an appropriate opportunity cost (*o*). When the summed consumer surpluses are measured in relation to earnings (*w*) a relative distributional effect (*V/w*) can be calculated. A Rawlsian variant of this (*R/w*) can be estimated by looking at the welfare gains of the lowest earning groups (for example, quantiles 1 and 2) in the earning distribution.

Finally, many of the attributes such as household income, distances, costs, visitation frequencies can be changed in order to simulate plausible scenarios affecting accessibility and equity concerns in ecosystem service provision. For example, scenarios can be articulated relating to the welfare and distributional outcomes of measures affecting the provision of ecosystem services, such as imposing entrance fees, improving access to beaches, incursion of residential development into national parks, provision of greater open spaces in cities etc. This comprises the fourth and final stage of the assessment framework (Fig. 1).

4. Data

The generation of synthetic microdata that underpins the evaluation of ecosystem services calls for extensive data. All data disaggregation is conducted on a national data set of census tracts (CT's) downscaled to the level of the individual household and geo-referenced to a dwelling unit. Three key data sources are:

- the Israeli Census of Population conducted by CBS (Central Bureau of Statistics) in 2008. This includes over 3000 CTs comprising over 2.3 million households and over 7.3 million inhabitants and is the only comprehensive source of socio-economic data at the resolution of the CT. We use aggregate CT level counts of households and household sizes, as well as population counts in order to create a disaggregated discrete dataset in which each household and each individual in a given CT is represented as a separate entity. All socio-economic attributes of households come from this source and are allocated using the IPF method (Section 3 above) to households and individuals. House prices and earnings come from other governmental sources (see Table 1) and are standardized to real 2009 values where necessary.
- a national buildings GIS layer comprising information on over 800,000 buildings comprising 1.4 dwelling units. This source includes variables on building height, number of floors, location, year built and number of dwelling units per building.
- a national road network GIS layer that yields information on road capacity and travel direction.

The original CT level data is disaggregated into households and individuals preserving the distribution of their attributes as represented in the census data. This process is comprised of three stages. In the first, a household (non-spatial) data set is created. The total number of households of each size (number of residents) is calculated using the total household count per CT and the distribution (in %) of household sizes in each CT. Each household is of a different size and all the households in each CT represent the marginal distribution of household sizes in the original census data. This is achieved by enforcing the automated IFP procedure of control total adjustment. The result is a dataset containing a unique representation of more than 2.3 million households nationally. The second stage involves creating an individuals' dataset. Households are further broken down to represent the number of residents in each household as distinct entities in the dataset. This dataset contains over 7.3 million entities representing individuals tied to (members of) households. Socio-economic attributes are allocated to each entity (households or individuals) in the third stage (as outlined in Section 3).The principle variables used in the allocation procedure, their sources and operationalization are outlined in Table 1.

5. Results

In order to estimate the distributional effects of changes in ecosystem service provision we compare hypothetical scenarios with baseline results. We estimate average household benefits nationally and then provide estimates for two localized scenarios with changes in ecosystem provision. The disaggregated nature of the data allows us to create benefit estimations at any spatial scale. We choose the Tel Aviv metropolitan area due to its combined demographic and economic dominance (60% of population and 55% of GDP on 12% of land area).

Table 1
Data description.

#	Variable	Aggregation level	Source	Allocation to	Categories	Data format
1	Total number of households	CT	CBS census 2008	Buildings		Total count in thou.
2	Household size (# of residents)	CT	CBS census 2008	Buildings & households	1, 2, 3, 4, 5, 6, 7 +	% of the total households
3	Total population	CT	CBS census 2008	Individuals		Total count in thou.
4	Number of children per household	CT	CBS census 2008	Households & individuals (age)	1, 2, 3, 4, 5 +	% of the total households
5	Senior citizens living on their own	CT	CBS census 2008	Households (size 1) Individuals (age)		
6	Age distribution	CT	CBS census 2008	Individuals	Age groups 0–17, 18–19, 20–29, 30–64, 65 +	% of the total population
7	Gender distribution by age	CT	CBS census 2008	Individuals	Age 0–17 m, age 0–17 f, age 18–64 m, age 18–64 f, age 65 + m, age 65 + f	% of the total population
8	Work force participation by gender	CT	CBS census 2008	Individuals	Age 15 + m, age 15 + f	% of the population above the age of 15 by gender
9	Occupation by gender	CT	CBS census 2008	Individuals	Academic and management (f, m); administration, sales and services (f, m); agriculture, industry and construction (f, m)	% of the population above the age of 15 by gender
11	Industry of employment by gender	CT	CBS census 2008	Individuals	Commerce and communications (f, m); public sector (f, m); agriculture, industry, infrastructure and construction (f, m); domestic services (f, m)	% of the population above the age of 15 by gender
12	Disabilities	CT	CBS census 2008	Individuals	National Social security categories	Aggregate score
13	Religion	CT	CBS census 2008	Individuals	Jewish, non-Jewish	% of the population
14	Year of immigration	CT	CBS census 2008	Individuals	Prior to 1960; 1961–1989; 1990–2001; 2002 +	% immigrants in the Jewish population by year of immigration
15	Education distribution by gender	CT	CBS census 2008	Individuals	Up to 8 years; 9–12 years; 13–15 years; 16 + years	% of the adult population by gender
16	Car ownership	CT	CBS census 2008	Households	0; 1; 2 +	% of the total households
17	Buildings	National GIS layer	Israel Land Survey	Households		Variables include, geometry, height, location
18	Earnings[a] from work	CT	National Social Security Institute 2009	Households, individuals	Quantiles as per national earning distribution	Quantiles, median
19	House prices	CT	National Tax Authority, 1998–2013	Households, individuals		Real 2009 values

[a] Household earnings used as a proxy for income.

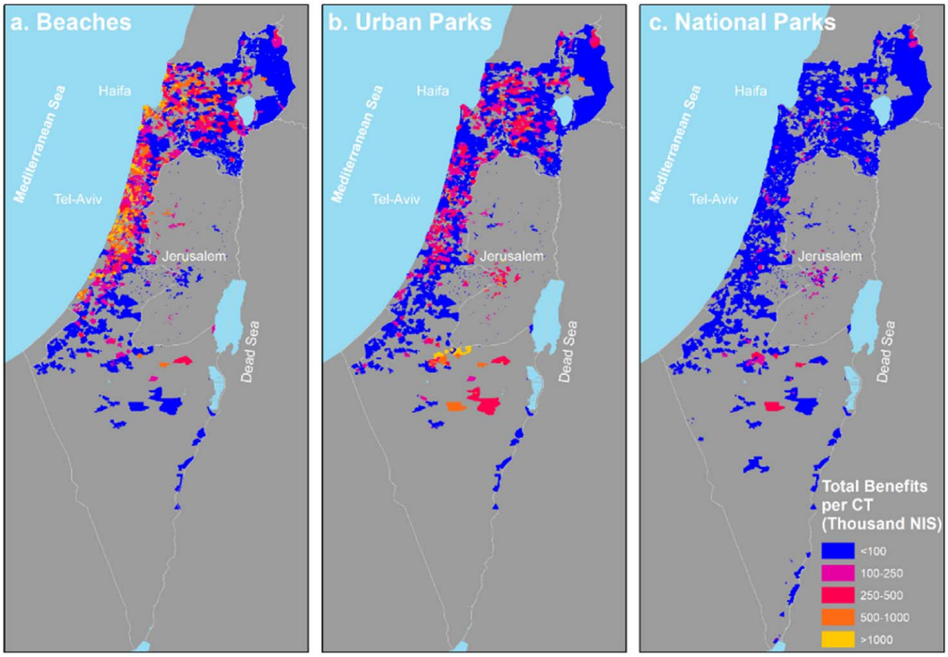

Fig. 3. Total value of household benefits from ecosystem services by type and census tract.

The two scenarios we chose to simulate relate to restraining access by imposing an entrance fee of 20 NIS[3] to Tel Aviv beaches and to the Yarkon Park, a large urban park in Tel Aviv. 20 NIS represents the average entrance fee to national parks in Israel.

In both cases, the distributional effects are captured by two outcomes. The first observes the effects in relation to earning quantiles and the second in relation to the distribution of benefits by house price quantiles. Due to lack of updated income data at the household level, we use earnings per person in the household as a proxy and henceforth refer to earnings from work per household member, as income. The value of house prices relate to the per square meter price for the dwelling unit. The household may own or rent the dwelling unit.

5.1. Baseline Results

Fig. 3 depicts total annual value of household benefits estimated nationally for each of the three types of ecosystem service featured here. As expected, the coastal distribution of population results in the shoreline sites generating the largest total benefits. Benefits ascribed to each category of ecosystem are contingent on distance to other ecosystems (see Appendix 2, Eq. (2)). Where accessibility to other ecosystems is difficult, compensatory benefits accrue to the ecosystem under consideration. That is, individuals can compensate for the loss of one type of recreational ecosystem services by choosing a different type (see Fleischer and Tsur, 2003 for further details). This is the case in the south of Israel where high benefits accrue from local parks in the absence of accessibility to beaches and to national parks.

Total household benefits from all types of site (i.e., the sum of the benefits depicted in Fig. 3) are represented in the panel b of Fig. 4. The national picture is one of two extremes. Coastal and northern parts of the Southern District capture the most benefits and the rest of the country captures very little. Representing the benefits as census tract averages, gives a more textured picture (panel a, Fig. 4).

Services supplied by the ecosystem services in this study are not traded through the market and thus individuals and households do not incur consumption-related costs. Under these conditions, public policy geared towards improving the welfare of low-income groups is expected to promote the progressive consumption of ecosystem services. This means that low-income groups should be Rawls-compensated for social deprivation by receiving more ecosystem services than those received by high-income groups. If this is the case, then the distribution of the ecosystem services should favor the poor.

Another issue to note is the role of house prices in this analysis. Although ecosystem services are not marketable, they still have an implicit market, i.e. the housing market. If a household wants to enjoy the ecosystem services of a beach or a park, it purchases a residence near these features. The premium on a house located near such an ecosystem measures, ceteris paribus, the cost incurred in receiving more ecosystem services than a household residing further away. Thus, if households in higher house price quantiles receive more ecosystem services we cannot contend that they receive them for free. They actually pay for them through the housing market.

As distributional effects relate to individuals, Fig. 5 shows baseline welfare effects in these units. The darker bars represent the value of the average per household consumer surplus from all three types of sites by income quantiles whereas the lighter bars represent the distribution of the same value by housing value quantiles. We would expect to find a clear correlation between both distributions. With respect to income, it is clear that quantile 5, with the highest income, receives the highest level of welfare. However, the result for the other quantiles is not definitive and all four quantiles elicit more or less the same level of welfare from the sites in this study. The picture is different when we look at the distribution of welfare by housing value quantiles. It is quite clear that households residing in high value housing elicit much more welfare from the sites consistently across all quantiles.

That seems to indicate that while these ecosystem services have no market value and are considered to be supplied free of charge, households implicitly pay for them via the housing market.

This lack of correspondence between income and house price quantiles may be due to the prevalence of a large rental sector in the Tel Aviv housing market. While nationally about one third of the Israeli households live in rental accommodation, in Tel Aviv, this share stands at 50% (CBS, 2016). This tends to smoke-screen the expected association between income and housing class as high income households can be found renting in non-commensurate accommodation. Had low-income households resided in low price houses we would have witnessed similar patterns of benefits across both distributions. However, the rental sector seems to iron-out some of the variance in this joint

[3] NIS = New Israeli Shekels; 3.6NIS = ~$1.

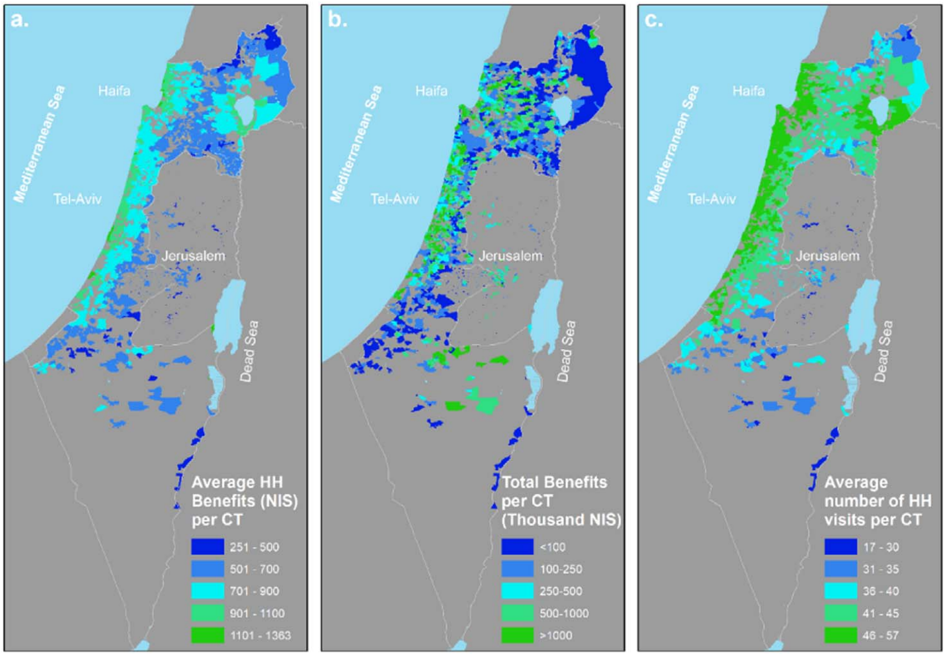

Fig. 4. Average and total household benefits and total household visits for all ecosystem services combined.

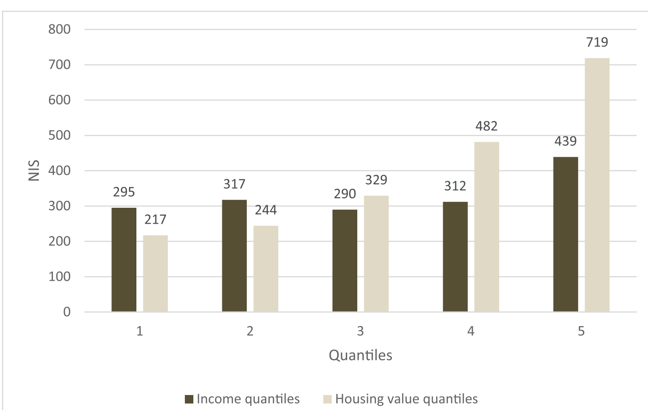

Fig. 5. Distribution of average consumer surplus per household member by income level (earnings per person) and house prices (price per square meter) (2015 NIS).

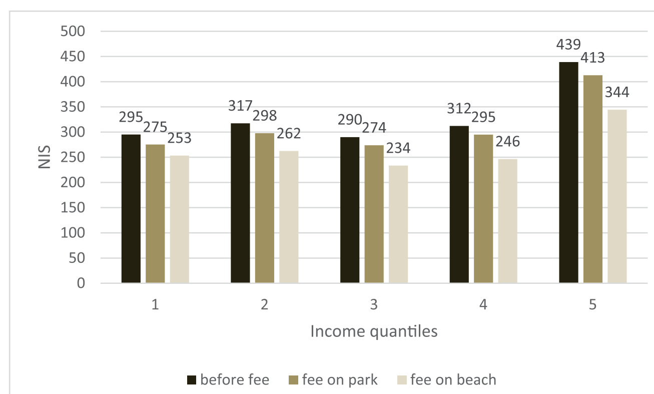

Fig. 6. Distribution of average consumer surplus per household member (2015 NIS) before and after imposing entrance fee on park and beach, by income groups.

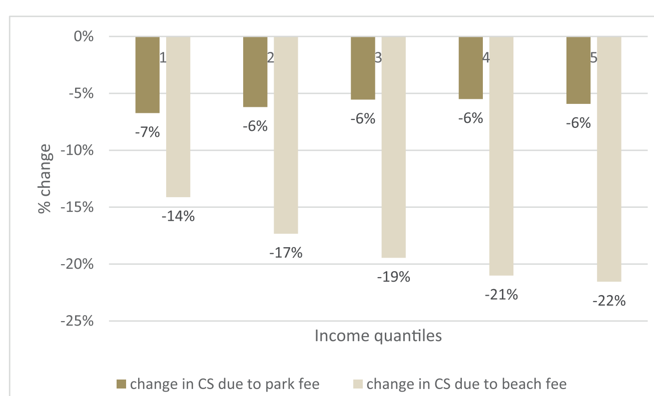

Fig. 7. Distribution of change in average consumer surplus per household member after imposing entrance fee on parks and beaches by income groups.

distribution. In addition, housing is spatially fixed with respect to the location of the ecosystem services. High priced housing is either close or distant from a given ecosystem with benefits distributed accordingly. Income classes however are not fixed in space. Poorer households can decide on the level of benefits they want from ecosystems and choose housing in line with these preferences.

5.2. Localized Scenarios

The welfare impacts of the two restraining entrance scenarios are depicted in absolute terms in Fig. 6 and relative (percentage change) in Fig. 7. Charging entrance fees for an ostensibly public good is a policy issue that arises periodically in Israel especially in the context of beaches. The main reason for charging a fee is the need for local authorities to cover maintenance costs via user fees. Accordingly, there is an ongoing debate about who incurs the burden of these fees. Some claim that they disadvantage lower income households who are requested to pay for a service they hitherto received for free. Others contend that high income households use these services to a greater degree and thus fees cause a progressive reduction in welfare. If this is the case, then imposing fees results in a more equitable distribution of welfare. Currently, some sites charge for parking and some for entrance. For example, the Yarkon Park in Tel Aviv charges a parking fee for non-residents. The two ecosystem services to be simulated are therefore closely related to the contemporary public discourse concerning open space management and represent likely future scenarios. We choose to apply the same simulation for both types of sites in order to compare between them.

Not surprisingly, both fees lead to a reduction in welfare (Fig. 6). The underlying intuition is that increasing the cost of visiting lowers the number of visits and the consumer surplus from each visit. This means that households not only visit less but since each visit now costs more, they elicit lower consumer surplus from each visit. The contribution of our analysis is the identification of the distributional impacts: who is most affected by these fees and are they applied progressively or regressively?

Figs. 6 and 7 show the changes in terms of consumer welfare. Fig. 6 shows that the absolute change in welfare resulting from imposing an entrance fee in Yarkon Park fluctuates around the same value for the lower four income quantiles, whereas the fifth income quantile suffers from the highest welfare reduction. However, Fig. 7 reveals that the percent change is about 6% across all income quantiles. That means that although households in the fifth income quantile suffer the highest reduction in welfare still there is no change in the relative position of all the quantiles. The entrance fee imposed on Tel Aviv beaches has a completely different impact. The absolute and relative loss of welfare increases across income quantiles. The fifth quantile loses 95 NIS compared to a 42 NIS loss to the first quantile. In relative terms, the fifth quantile suffers a 22% welfare reduction compared to 14% for the first quantile. It should be noted that these results are in terms of consumer surplus elicited from the ecosystems. Nevertheless, when we calculate the percent change of the consumer surplus relative to income level, the changes are very small and regressive. This means that the poorer quantiles lose a higher percentage of their income than the rich households.

Figs. 8 and 9 show the distributive effect of beach and park entrance fees across house price quantiles. Here also we see the lack of congruence in the distribution of welfare between the income and house price quantiles. In absolute terms, higher house price quantiles consistently lose more welfare than lower price quantiles for both type of ecosystem service charges. However, in relative terms we obtain different distributive effects. Fig. 9 reveals that imposing fees on park entrance (restricting access) affects the first (lowest) house price quantile relatively less than the higher-order quantiles, eliciting a Rawlsian effect. Imposing fees on beach entrance however has a regressive relative distributional impact. It generates a much higher welfare reduction than the park fee and this is felt more acutely by lower housing classes (− 21%) than by higher housing classes (− 16%).

The methodology illustrated here allows us to compare two ostensibly similar policy measures related to the management of ecosystems. We show that despite imposing the same entrance fee on an urban park and an urban beach, the effect on welfare implicit in these ecosystems is completely different. The charges do not only induce different reductions in welfare levels but they also generate different distributional

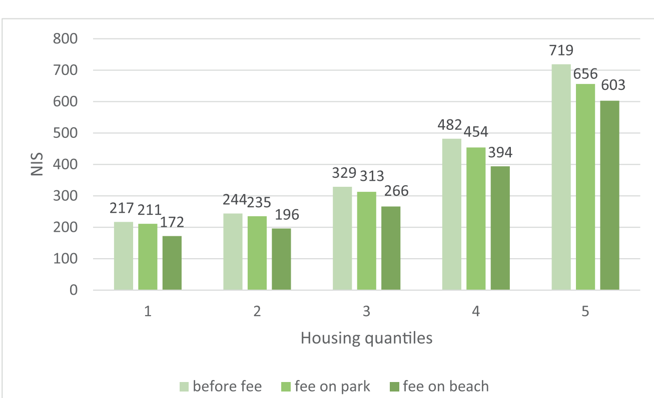

Fig. 8. Distribution of average consumer surplus per household member before and after imposing entrance fee on parks and beaches by housing classes.

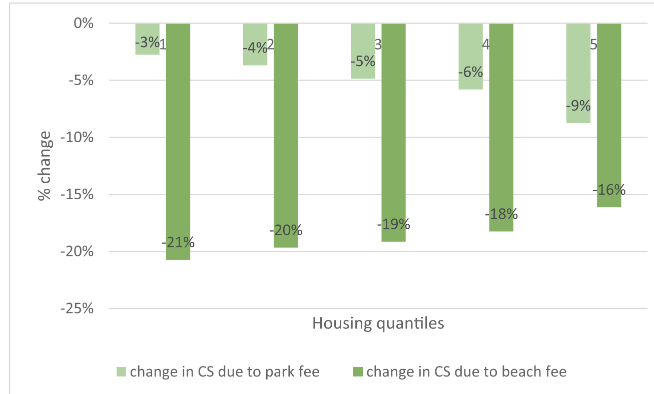

Fig. 9. Distribution of change in average consumer surplus per household member after imposing entrance fee on parks and beaches by housing classes.

effects. Beach fees alter the distriubution of benefits leading to a more Rawlsian distruibution of ecosystem services whereas the urban parks fees have no distributional effect.

We also show that measuring the welfare impacts of reduced accessibility to ecosystem services seems contingent on the type of distributional outcome chosen. One possible explanation for this is the fixed location argument evoked above. High price residences are located proximate to beaches with few ecosystem- consumption alternatives. Households are therefore less sensitive to changes in the price of a beach visit since this is the closest (and only) recreational possibility. In contrast, lower housing classes substitute the urban park under consideration with other less costly alternatives. Another explanation for the lack of similarity across distributional outcomes is the distorting effect of the extensive local rental market. This results in households from low income quantiles choosing to occupy residences in high house- price quantiles (students and migrants doubling up, etc.).

6. Conclusions

Despite the recent accumulation of ecosystem service evaluations, these have largely over-looked the distributional effects arising from the provision of ecosystem services and their likelihood of generating Rawlsian-type outcomes. Existing studies emphasize deterioration in the supply of ecosystem services mainly because they have no market value and thus are largely ignored in the planning and management process of ecosystems. With increasing awareness of the importance of ecosystem services and their contribution to national and local welfare, there is a growing need to evaluate who benefits from these services. We present a methodology which enables researchers, policy makers and planners to understand the distributional effects of changes in ecosystem services.

The two scenarios we simulate relate to restraining entrance to a large park in Tel Aviv and to the beaches of Tel Aviv. However, the resultant effects on households are very different, both absolutely and relatively. Our results show that although the ecosystem services are similar (recreational benefits from open space) and a uniform fee is charged, the outcomes diverge greatly. This is because the number of visits to the sites and the elicited welfare depend on the type of site, distance from the site and on household attributes. Thus, changes in the value of the welfare depend on these variables. Implicitly ignoring them means that we cannot evaluate how they differentially affect households. In our simulation, we illustrate that constraining accessibility to beach ecosystem services causes a greater reduction in welfare than constraints on access to park ecosystem services. In terms of consumer surplus, high income households lose relatively more in welfare terms than low income households from the imposition of a beach fee—a progressive outcome. However, the loss of welfare relative to income is higher for low-income households than for high income classes. The outcome in terms of consumer surplus is reversed when measured in terms of housing classes: low price dwellings (and their residents) suffer more from restricted access to beaches than do high-price dwellings. Constraining access to urban parks has little variation in terms of income distribution effects perhaps due to the ubiquitous nature of urban parks and the existence of alternatives. But in terms of housing classes the distributive effect is regressive. The fact that ecosystem surpluses are capitalized into house prices can have serious implications for public policies aimed at social mixing and generating affordable units in new housing developments proximate to ecosystem services.

Our method has harnessed the tools of micro-simulation and GIS-based analysis to calculate ecosystem benefits at a level of spatial resolution and accuracy hitherto attempted. We address the distributional outcomes of policy measures often used with respect to pricing benefits from ecosystem services. As such, both the method and substance of this work are likely to be of interest to planners and decision makers. We illustrate that it is possible to go beyond the estimation of total change in ecosystem services and to disaggregate to the level of the individual household. Given this level of spatial detail, evaluations of ecosystem services can be recreated at any level of geographic granularity necessary.

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Appendix 1. The Spatial Allocation Process

In recent work (Felsenstein et al., 2016) we have grounded spatial allocation in a 'reverse' Alonso-type mechanism where individuals are allocated according to preferences and budget. In this way we back-out spatial preferences from rents. Note q = land size, t = proximity to location, $p(t)$ = land price function from point t, $k(t)$ = commuting expenses function from t, Pz = price of compound good Z (see Alonso, 1964, pp. 21, 31).

Assuming individuals within polygon P confirm to classical economic behavioral assumptions, and assuming a representative Cobb-Douglas utility function:

$$U = q^{\alpha} \cdot t^{\beta} \cdot Z^{\gamma}$$

This allows for the representation of an individual's willingness to tradeoff location and land area as $\frac{\alpha}{\beta}$. The median and extreme boundaries of this preference ratio are deduced from the price structure of P, recognizing that these characteristics must allow for the given price structure.

We denote household (H) tradeoff as: $H_{\frac{\alpha}{\beta}} = f\left(H, Norm\left(\frac{\overline{\alpha}}{\beta} + S, \frac{\alpha}{\beta}_{Range}\right)\right)$, using an allocation function.

After developing Alonso's results for an individual's maximization of utility, it is possible to state the maxima condition as:

$$\min_{j, for\ i}\left[\frac{\alpha}{\beta_i} - e_{t_j, p(t_j)} + \frac{k(t_j)}{p(t_j)} \cdot e_{t_j, k(t_j)}\right] \rightarrow \min_{j, for\ i}\left[\frac{\alpha}{\beta_i} - \frac{t_j \cdot q_j}{dI - k(t_j)}\left(\frac{\partial p}{\partial t} + \frac{\partial k}{\partial t}\right)\right]$$

$$s.\ t. \quad 1)\ dI = P(t)q + K(t), \qquad\qquad s.\ t. \quad j \in P$$

$$2)\ j \in P$$

where j is the array of available dwelling units. Individuals test all locations and choose the closest to their preferences and satisfying their budget. Within P, households attempt to minimize the difference between their given $\frac{\alpha}{\beta_i}$ and the location's implications in terms of price. A household that records a zero difference maximizes location in accordance with its preferences.

Appendix 2. Consumer Surplus From Visiting Sites Representing Different Ecosystems

Following Fleischer and Tsur (2003) consumer surplus is derived as follows:

$$U_{mj} = v_{mj} + x_{mj}{}^{o}x_{mj}b - rc_{mj} + x_{mj};\ m = 1, 2, \ldots, M;\ j = 1, 2, \ldots, J_m \tag{1}$$

represents the utility an individual derives from visiting site j in ecosystem type m, where x_{mj} is a vector of individual site characteristics, c_{mj} is the travel cost, v_{mj} is the deterministic part of the utility, β and ρ are parameters and J_m the number of sites in ecosystem type m. The ξ_{mj}'s are assumed to be extreme value random variates, uncorrelated between groups but correlated within groups, with the inclusive coefficient γ representing the degree of correlation within each group.

Denoting the index set and the number of sites in group m by I_m and J_m respectively, $m = 1, 2, \ldots, M$, the utility an individual derives from visiting (a site in) ecosystem type m is:

$$U_m = Max_{j \in I_m}\{u_{mj}\} = V_m + \gamma \ln J_m + \gamma \ln H_m + \varepsilon_m, \quad m = 1, 2, \ldots, M, \tag{2}$$

where

$V_m = (1/J_m)\Sigma_{j \in I_m} v_{mj}$ is the average (deterministic) utility of ecosystem type m,

$$H_m = (1/J_m)\sum_{j \in I_m} e^{(v_{mj} - V_m)} \tag{3}$$

is a measure of the heterogeneity of sites in ecosystem type m, γ is the inclusive coefficient defined above and the ε_m, $m = 1, 2, \ldots, M$, are iid extreme value variates.

The probability of visiting ecosystem type m is

$$P_m = \frac{\exp(V_m + \gamma \ln J_m + \gamma \ln H_m)}{\sum_{m=1}^{M}\exp(V_m + \gamma \ln J_m + \gamma \ln H_m)} \tag{4}$$

and the per trip consumer surplus:

$$CS = \frac{1}{\rho}\ln\left\{\sum_{m=1}^{M}\exp(V_m + \gamma\ln J_m + \gamma\ln H_m)\right\}, \tag{5}$$

The consumer surplus due to ecosystem l is

$$CS_l = CS - CS_{-l}, \tag{6}$$

where

$$CS_{-l} = \frac{1}{\rho}\ln\left\{\sum_{\substack{m=1\\ m\neq l}}^{M}\exp(V_m + \gamma\ln J_m + \gamma\ln H_m)\right\} \tag{7}$$

represents the surplus index in the absence of ecosystem l.

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