How much is that birdie in my backyard? A cross-continental economic valuation of native urban songbirds

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Abstract Human-wildlife interactions in urban areas, both positive and negative, often involve people and birds. We assess the economic value placed on interactions with common native songbirds in two different urban areas (Berlin, Germany and Seattle, Washington, USA) by combining a revealed preference (recalled expenditures on bird feed) and a stated preference approach (determining willingness to pay for conservation or reduction of birds). Residents in both cities purchase bird food, engage in a range of bird-supporting activities and are generally willing to pay a small amount for native songbird conservation. Demographic, cultural and socio-economic factors, as well as specific attitudes towards birds and general attitudes about conservation were found to influence these decisions. This study presents the first attempt at estimating the economic value of enjoying common native urban songbirds and estimates the lower bound to be about 120 million USD/year in Seattle and 70 million USD/year in Berlin.

Keywords Birds · Conservation · Human surveys · Urban wildlife · Willingness to pay

Introduction

In the past, wildlife conservation has focused on natural areas or reserves. While these efforts are important, rapid urbanization globally is increasing the need of conservation in urban areas (e.g., Dunn et al 2006). Drastic changes in landcover, vegetation and habitat quality are all factors that affect native wildlife communities sending some species toward local extinction but also promoting the spread of some adaptable species (Marzluff 2001). Nevertheless, it is not only changes in the physical environment that influence urban species. Humans and their

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actions can influence community structure, population dynamics and behavior of wildlife (Kinzig et al. 2005; Fuller et al. 2008).

Birds are often the most prevalent and visible animals in urban areas and present the greatest opportunity for humans to interact with wildlife (US Department of Interior et al. 2011). Humans provide attractants such as bird feeders, nestboxes, and water sources that influence the abundance and diversity of birds (Fuller et al. 2008; Robb et al. 2008; Clucas et al. 2011). In general human well-being is influenced by a view of nature (Maller et al. 2005; Jorgensen and Gobster 2010), and the presence of birds and greater bird diversity positively influences human perception of green areas (e.g., Fuller et al. 2007) as well as enhancing life satisfaction (Rehdanz 2007). Thus, an increase in the presence and diversity of birds in urban areas has the potential to provide a service that benefits humans directly.

Human efforts to increase the abundance of desirable birds (e.g., aesthetically pleasing songbirds) or to decrease the abundance of birds they consider a nuisance can take both private and public forms. In this paper, we assess both by focusing on monetized (bird food expenditure) and non-monetized (a count of activities engaged in to attract birds) private actions, and also the willingness to pay (WTP) for publicly financed actions aimed at influencing bird abundance. It is important to focus on both private and public actions because of the mixed economic nature of the service that birds provide. Urban birds can be thought of as a public good because more than one person can enjoy interactions with the same bird (i.e., enjoyment is not fully rival) and a person can enjoy birds without contributing anything to feeding them or improving their habitat (i.e., enjoyment is not fully excludable). Moreover, individuals that do not engage in private bird-supporting activities may still be willing to pay for public actions to enhance their interaction with birds (Hamilton et al. 2003).

As a result of the complex nature of the aesthetic services birds provide to humans, both revealed preference (observed private actions) and stated preference (hypothetical behavior in supporting public actions) approaches are necessary to capture the full economic value of enjoying the presence of urban birds (as done e.g. by Becker et al. 2010 in a context of an endangered bird valuation). To date, significant research effort has been devoted to studying the economic value of rare and endangered birds (Loomis and White 1996, Richardson and Loomis 2009, Booth et al. 2011), with some effort going to valuing more common birds but in unique settings such as swallow roosts (Clark 1987) or stork villages (Czajkowski et al. 2014). Bird-centered tourism ("avitourism") is justifiably a very active area of research, including economic valuation studies (Czajkowski et al. 2014 provide an excellent background). We wish to consider a complementary question of economic value of common urban birds. To our knowledge, there have not been any studies assessing the economic value of urban bird enjoyment (but see DeGraaf and Payne 1975). Furthermore, the ability of private actions to enhance human-bird interactions is clearly a distinguishing feature of urban wildlife and contrasts with the more traditional focus on wildlife in natural areas, where public action is often necessary in order to provide the good (as in, e.g., Stevens et al. 1991, Loomis and White 1997, and in studies reviewed by Richardson and Loomis 2009). We aim to fill this gap by providing a lower monetary bound of the total economic value that Berlin, Germany and Seattle, USA, residents place on interactions with common, native songbirds.

There are several reasons why we speak of the lower bound of the total value. First, looking at consumer expenditure underestimates total economic benefit of any good because it ignores the fact that some consumers are willing to pay more than what they had to pay in the market. A full accounting would require estimating the entire market demand. Second, many bird-supporting activities are not easily monetized. We compromise by monetizing one component expected to represent a significant portion of those activities (expenditure on bird food) and by constructing a non-monetized count index of bird-supporting activities. Finally, we monetize

stated preferences for common native songbirds with a WTP analysis for typically pleasing (finches) and displeasing (corvids) species.

Human participation in bird-supporting activities and their WTP to change urban bird abundance can be influenced by many factors. For instance, age, gender, culture, education, economic status and population density affect bird feeding (Lepczyk et al. 2004; Fuller et al. 2008; Clucas and Marzluff 2012) and WTP for conservation action can be affected by age and income level (Kaval and Roskruge 2009). Therefore, we assess the value people place on urban bird enjoyment by surveying residents in Berlin, Germany and Seattle, Washington, USA across neighborhoods of known varying levels of human densities and socio-economic status. We first hypothesize that older, higher socio-economic status respondents in both cities will spend most on birds (e.g., Fuller et al. 2008) and that Germans will be most likely to engage in bird-supporting activities (e.g., Kellert 1994). Second, we hypothesize that WTP will be influenced by attitudes towards birds and general concern for conservation (Tisdell et al. 2005) in addition to age, socio-economic status (Kaval and Roskruge 2009), and culture (Kellert 1994). We also provide a comparison of the effects of these factors across the two cities and across private and public contributions to enhancing human experience with common urban birds.

Methods

Study areas

We conducted our surveys in two comparable cities (in terms of urbanization, population density and latitude) that have been the focus of a long-term urban ecology study (Clucas et al. 2011). These cities differ in history, culture and, pertinent to our study, general attitudes towards wildlife (see Kellert 1994). Seattle, Washington, USA (47°36′35′N, 122°19′59′W) was originally settled by persons of European descent around 1850. Currently, the Seattle metropolitan area has a population size of 2.58 million adults and 1.48 million households (U.S. Census Bureau, 2011). Berlin, Germany (52°30′2′N, 13°23′56′E) was originally settled by Slavic tribes around 720 but it was not until 1244 that the city of Berlin was founded. Currently, the Berlin metro area has a population size of 2.97 million adults and 1.99 million households (Amt für Statistik Berlin-Brandenburg, 2010).

Study sites

We had 8 study sites in Seattle and 10 in Berlin (approximately 0.2 to 3 km^2). In each city, we selected two replicate study sites across a wide geographic gradient consisting of four levels of varying population densities: (1) urban (city center, apartments, 252 ± 143 [average residents/ha$\pm$SD]); (2) dense suburban (detached family housing, 46 ± 23); (3) light suburban (detached family housing, 24 ± 17); and (4) rural (villages, farms, detached family housing, 8.5 ± 5; for more details see Clucas et al. 2011, Clucas and Marzluff 2012). Within the dense suburban and light suburban levels where it was most distinguishable, we selected one relatively high median household income site (e.g., 70,000–100,000 USD) and one medium median household income site (e.g., 40,000–53,000 USD) based on census data (U.S. Census Bureau 2000; FIS Broker 2008) to get a range of income levels in suburban areas. The two additional sites in Berlin represented large high-rise apartment communities in the outer zone of the city: Gropiusstadt and Märkisches Viertel (built in the 1960–70s as large housing communities). These sites have urban-like population densities but lie outside the city center. We did not sample similar sites in Seattle as large housing communities of such nature do not exist.

Human surveys

We conducted our human surveys during weekdays (1000 to 1900 hours) in Berlin from August 2008 to December 2008 and in Seattle from October 2009 to February 2010. We used face-to-face interview style surveys because they allow for a longer survey with more in depth and complex questions de Leeuw 1992 and because certain sections of the survey involved showing photographs to the respondents. The surveyor read the questions out loud, filling out the survey and answering any questions the respondent had during the interview. Households were surveyed door-to-door within the study site until 50 surveys were completed. If necessary, due to low first attempt response rate (as was the case in Seattle sites and a few Berlin sites) households were attempted a second time, which resulted in 25–30 surveys completed. Success rate (residents agreeing to complete survey) was recorded during the Seattle surveys (Online Resource 1).

   Residents were briefly told about the focus of the survey (human-bird interactions) and that no specific knowledge of birds was necessary. The survey then consisted of three sections and most of the questions used in our present analysis were closed-ended, requiring a categorical response (see Online Resource 2 for the complete survey instrument).

1. Demographic questions: a) gender, b) age, c) housing ownership status (rent or own), d) employment status, e) education level, and f) income level.
2. General questions about attitudes and actions towards birds and conservation: a) How often do you notice birds around your home, b) Do you provide food for birds, c) How much money do you spend annually on bird food (we asked residents to recall monthly bird food expenditure and how many months of the year they fed, which were subsequently converted to annual values), d) Do you provide nestboxes for birds, e) Do you provide plants to specifically attract birds f) Do you do anything else to attract birds (e.g., birdbaths), g) Are you bothered by the noise birds make around your home, h) How important are environmental or conservation issues to you, and i) Do you belong to any organizations that support bird conservation?
3. Willingness to pay for common urban songbird questions: We selected two types of bird species to study the WTP for public programs to change the abundance of birds: finches and corvids. In Seattle, we chose the house finch, *Carpodacus mexicanus*, and the American crow, *Corvus brachyrhynchos*; in Berlin, the greenfinch, *Carduelis chloris*, the hooded crow, *C. cornix*, and the European magpie, *Pica pica*). We selected these species for several reasons. First, all of these species are common and are found at each of our study sites. Second, people tend to perceive corvids and finches differently. Corvid species tend to evoke negative responses from people and they are sometimes viewed as a pest species and their appearance and vocalizations are often disliked. In contrast, finches are colorful and have pleasant sounding appearance and vocalizations, which evoke more positive (or neutral) responses. Thus we expect respondents to be more willing to pay for finch than corvid conservation (and possibly even to have a positive WTP to reduce the abundance of corvids).

   We presented color photographs of finches and corvids (in random order for each participant, in order to eliminate any order effects). For those reporting a positive (negative) attitude toward a bird species, we asked for a WTP to increase (reduce) the abundance of species. We asked both WTP questions for those with neutral attitudes. This technique has been used in wildlife valuation contexts where researcher's expectation regarding the preference for or against wildlife is not clear *ex ante* (e.g., Duffield 1991).

In Berlin, if the participant said "no" to a general willingness to pay question, the questioning stopped for that species. If they said "yes", the participant was then asked if they would be willing to pay specific monetary amounts. We used the double-bounded dichotomous choice (DBDC) elicitation format (Cameron and Quiggin 1994). The interviewer selected the amounts randomly from a list of eight WTP (bid) sets, with each set represented by three amounts: low, middle and high (two examples being: 15, 60, 120 and 45, 90, 150 United States Dollar [USD]). The first amount asked was the middle amount (e.g., 60 USD) and if the participant said yes, she/he was then asked the high amount (e.g., 120 USD); if no, the low amount (e.g., 15 USD; see Online Resource 2). The amounts presented to the respondents were selected using the results of a Berlin pre-test. In Seattle, survey participants were immediately asked if they would be willing to pay the middle WTP amount in a randomly chosen set. As a result, we have conditional WTP responses for Berlin (conditional on the respondent stating a non-zero WTP), and an unconditional willingness to pay for Seattle. We correct for this conditioning in comparing our estimates using the procedure outlined in Haab and McConnell (2003), and we assume that Berlin respondents who answered "no" to both willing to pay to increase or to reduce bird populations are indifferent (have zero WTP). Specifically, to ensure the two cities' WTP estimates are comparable we follow Haab and McConnell (2003) and compute $WTP=0 \cdot Pr\ (WTP=0)+(WTP)^{hat} \cdot Pr\ (WTP\neq 0)$ to account for the share of population that has a non-zero WTP. We analyze finches and corvids separately, thus estimating four WTP models, varying by city and bird species. Note that we use the Euro-dollar exchange rate for November 2010 (1.35 USD to 1 Euro) and report results in U.S. dollars.

Data analysis

For the monetized bird food expenditure responses, we used the censored regression (Tobit; Greene 2003) model to account for a large proportion of zero expenditure responses. The annual recall-based bird food expenditure (in USD) was regressed on the set of neighborhood, attitude, and socio-economic characteristics. For the non-monetized index of bird-supporting activities , we created a count variable (ranging from 0 to 4), by summing the indicators of providing (a) bird food, (b) nestboxes, (c) specific vegetation, and (d) other (e.g., bird baths). We use the zero-inflated Poisson (ZIP) hurdle model (Haab and McConnell 2003; Greene 2003) to account for the count structure of the response variable, for the large number of zero values of the activity index, and for potentially varying factors influencing the decision to engage in *any* activity (participation equation), and the decision on the number of activities reported (intensity equation).

In the DBDC elicitation format, based on survey responses to WTP amounts, a respondent's WTP is placed in a particular interval of values. Thus, response data is interval-censored (Haab and McConnell 2003), and interval regression was used. All models were estimated by maximum likelihood using SAS System version 9.2 (SAS Institute 2010).

Independent variables: Based on existing literature, we postulate that a set of neighborhood, attitudinal, socioeconomic, and demographic factors may affect the bird food expenditure, the number of bird-supporting activities, and the individual WTP. We used population density to capture neighborhood characteristics and membership in conservation organizations and an indicator of conservation attitude (equal to 1 if conservation/environmental issues were rated "most" or "very important" relative to other issues) to capture conservation attitudes. To capture attitudes towards birds, we used the indicators whether respondents noticed birds (equaling 1 if reporting noticing birds at least weekly) or were bothered (1 if "bothered very much" or "somewhat bothered") by birds. Socioeconomic and demographic factors included

age, gender, ownership of residence, employment status (full time or not), income, and being born in a foreign country. We explored the full set of these variables but report results based on more parsimonious models selected using likelihood ratio tests.

Aggregation of expenditures and willingness to pay estimates based on survey samples

It should be noted that the original sampling strategy (stratified along the population density gradient) was designed to look for differences across the different urban forms in Seattle and Berlin and was not designed to be an accurate representation of the two cities' populations. Therefore, aggregating expenditure values or the willingness to pay estimates for the two cities based on uncorrected samples would likely lead to bias. However, human populations in both cities are studied extensively by government statistics agencies in both countries. As a result, we can employ a post-stratification (e.g., Cochran 1977) approach to construct weights for sample observations using auxiliary census information. To correct for non-representativeness of the sample based on socio-demographic information, we construct post-stratification weights as a ratio of percentage of specific category of respondents in the population to the percentage of a specific category of respondents in the sample (e.g., Little 2007). To correct for the stratification based on population density (and in order to aggregate expenditure and willingness to pay values at the household level), we similarly construct the weights using official estimates of numbers of households in different population density categories. The weight used in aggregation was constructed as a product of these two weights (Online Resource 1). Valuation and level of activity presented in the results utilize the weighted sample unless otherwise indicated (unweighted sample results were quite similar and are available upon request). In addition, we use respondent (individual) characteristics to estimate the models and to construct post-stratification weights but aggregate the expenditures and willingness to pay results at the level of the household, as both bird food expenditure and paying a tax for conservation (the payment vehicle in the stated preference question) is best thought of as occurring at the level of a household.

Results

General survey responses

We successfully surveyed 209 residents across 8 sites in Seattle and 460 residents across 10 sites in Berlin. In Seattle, the response rate (residents asked that agreed to complete the survey) was 69.1% (209/304), however, 57.7% of the time no one answered the door, thus including these "no answers", the response rate was 28.9% (see Online Resource 1).

In unweighted samples (Online Resource 1), there were no significant differences in the number of males and females surveyed in either city although participants tended to be female (60%) and those sampled in Berlin were slightly older than those in Seattle (mean±SE: Seattle: 49.64 ± 1.97 years, Berlin: 55.17 ± 0.96; t-test: t_{665}=−3.66, $p < 0.001$). Seattle respondents were more likely to own their homes (78%) than Berliners (48%; X^2_1 = 52.53, $p < 0.0001$), have a higher median annual income (Seattle median: 75,000–100,000 USD; Berlin median: 16,250–24,300 USD; Mann-Whitney test: $U = 10{,}991.5$, $P < 0.001$), have a greater number of years of education (Seattle: 15.74 ± 0.13 years, Berlin: 12.05 ± 0.13; t-test: t_{656} = 17.6, $p < 0.001$), and more likely to be employed full time (Seattle: 31%, Berlin: 18%; Mann-Whitney test: $U = 41{,}379$, $P < 0.001$). In both cities, the majority of respondents noticed birds on a weekly basis (99%), and most respondents were not bothered by noise that birds make (Berlin:

86%, Seattle: 92%). Berliners placed a higher importance on conservation and environmental issues (80.0% responded these issues were very or most important) than Seattle respondents (57.4%; $X^2_3=38.39$, p<0.001); however, Seattle respondents were more likely to belong to a conservation organization than those in the Berlin sample (Seattle: 17.7%, Berlin: 4.8%; $X^2_1=$ 29.84, p<0.001).

Revealed preference results: Expenditure on bird food

Although more respondents in Berlin report purchasing bird food, Seattleites spent more money annually on bird food (mean +/-SD: 49.17+/−146.26, range: 0–1,200 USD) than Berliners (11.26+/−23.87, 0–224 USD). Weighting our results to account for sampling bias produced similar results (Seattle: 52.26+/−175.37 USD; Berlin: 11.37+/−19.94 USD). Nevertheless, in terms of expenditure relative to household incomes this difference is not as large (approximately 0.052% for Seattle and 0.047% for Berlin).

In Seattle, expenditure increased with age, education, and positive conservation attitude, and declined with increasing population density (Table 1). Similarly, expenditure in Berlin increased with age and conservation attitude and decreased with increasing population density (Table 2). In contrast however, education had a negative effect on expenditure in Berlin and expenditure also increased with noticing birds on a weekly basis, belonging to conservation organization and increasing income level.

To aggregate expenditures based on the recalled expenditure models, we used the sample mean of Tobit model predictions (using post-stratification weights). Unlike an ordinary least squares regression, the prediction in the censored regression is a highly nonlinear function of the data and parameter estimates (Greene 2003). Accounting for censoring, mean annual predicted expenditure is 11.75 USD (sample SD=7.26) in Berlin and 72.30 USD (sample SD=93.35) for Seattle. Note that these are not identical to the raw data means. This is to be expected when a correction for censoring at zero is employed (see, e.g., Cameron and Trivedi 2005), and when the estimated error standard deviation is large (as it is in the Seattle sample).

To quantify estimation uncertainty in our expenditure estimates, we employ the Krinsky-Robb procedure (Haab and McConnell 2003) to simulate the confidence intervals around expected expenditure estimates implied by the parameters estimated using maximum likelihood. For the mean expected annual household expenditure, the 95% confidence interval is (11.55, 12.24) in Berlin and (67.75, 82.53) in Seattle. Using mean model-predicted expenditures for the post-stratified samples, and using official household counts for both cities, we

Table 1 Seattle Tobit model estimates of annual food expenditure. Note: in terms of interpreting the coefficients, negative coefficients are interpreted as having a negative impact on expenditure ($n=158$; degrees of freedom = 1; log-likelihood $=-353.65$)

Variable description	Estimate	Standard Error	t Value	Approximate Pr>|t|
Intercept	−1,951.03	0.45	−4,274.6	<0.0001
Age	42.57	5.10	8.34	<0.0001
Population density	−2.35	0.79	−2.98	0.0029
Positive conservation attitude	161.37	67.06	2.41	0.0161
Years of education	89.47	8.26	10.83	<0.0001
Age*Education interaction	−2.23	0.34	−6.58	<0.0001
(error standard deviation)	293.06	33.83	8.66	<0.0001

Table 2 Berlin Tobit model estimates of annual food expenditure. Note: in terms of interpreting the coefficients, negative coefficients are interpreted as having a negative impact on expenditure ($n= 448$; degrees of freedom =1; log-likelihood =−982.20)

Variable description	Estimate	Standard Error	t Value	Approx Pr>|t|
Intercept	293.06	33.83	8.66	<0.0001
Notice birds at least weekly	141.54	8.90	15.90	<0.0001
Age	0.48	0.16	2.96	0.0031
Population density	−0.13	0.02	−5.19	<0.0001
Positive conservation attitude	19.80	7.26	2.73	0.0064
Conservation organization membership	25.84	11.42	2.26	0.0237
Years of education	−1.96	0.99	−1.97	0.0487
Income category	3.25	2.08	1.56	0.118
(error standard deviation)	44.69	2.73	16.34	<0.0001

estimate that total annual expenditures are 72.30*1.48=107.0 million USD/year for Seattle (95% CI: (100.3,122.1)) and 11.75*1.99=23.4 million USD/year in Berlin (95% CI: (23.0, 24.4)).

Revealed preference results: Nonmonetized bird-supporting activities

We considered two decisions for nonmonetized activities: (1) the participation decision (to engage in any bird-supporting activity) and (2), conditional on deciding to participate, the intensity decisions (how many activities to engage in). Overall, Berliners were slightly more likely to participate in any activities than Seattlelites (58.7% versus 53.4%, respectively), however this difference was not significant ($X^2_1=$ 1.53, p=0.2) and the mean predicted participation levels in bird-supporting activities (using weighted samples) were similar in Berlin and Seattle (1.17+/−0.68 and 1.03+/ −0.63 activities, respectively).

In Seattle, the likelihood to participate in bird-supporting activities increased with positive conservation attitude and home ownership and decreased with increasing population density and full time employment (Online Resource 1). However, within the subset of residents participating in least one activity, females, older residents and those that were employed full time engaged in more activities.

Similar to Seattle, Berlin residents were more likely to participate in bird-supporting activities if they had a positive conservation attitude and owned their home and less likely in population dense areas (Online Resource 1). However, Berliner participation was also positively influenced by age and years of education and negatively influenced by being bothered by birds. In addition, there was a negative interaction of age and education suggesting younger residents (with less education) participate less. Within the subset of residents participating in least one activity, positive conservation attitude, home ownership and income increased the number of activities and greater population density decreased them. Surprisingly, those bothered by birds also engaged in more activities (perhaps because they are bothered by some species but attempt to attract other species they find appealing). Also conditional on deciding to support birds, more educated respondents engaged in fewer activities, with the impact moderated somewhat for older residents (positive coefficient on age-education interaction).

Stated preference results: Willingness to pay (WTP)

We estimate the following WTP values for finches and corvids using the post-stratification weighted samples and a Krinsky-Robb procedure to calculate 95% confidence intervals (CI) to address uncertainty in these estimates. For finches in Seattle, the mean expected WTP to increase abundance of finches was 19.96 USD/year (SD=38.30; CI=(18.07, 23.48)). For Berlin, the mean expected WTP to increase finches was 81.03 USD/year (SD=14.77; CI=(78.25, 83.32)). Recall, however, that our Berlin WTP estimates are conditional on the respondent stating they were willing to pay some positive amount for public bird projects. Nineteen and a half percent of Berlin respondents stated that, so we corrected the expected WTP (Haab and McConnell 2003) to be 0.195*81.03=15.80 USD/year (CI: (15.26,16.25)).[1] As a result, while mean expected WTP in both cities are quite close, Seattle has a higher WTP for finches (Fig. 1).

The two cities differed in their preferences for corvids. For Seattle, sample mean expected WTP is negative because the estimate is 8.43 USD/year (SD=25.33; CI=(0.54, 16.33)) to decrease abundance of crows. Although, on average, the negative preference toward crows in Seattle was significant, actually reducing crow abundance would likely be a controversial issue, as those respondents who belong to environmental organizations had a significant (at 5% level) positive WTP to increase crow abundance (CI: (0.52, 93.91)). For Berlin, among those who were willing to pay some amount for a public program, the mean expected WTP to increase the abundance of crows and magpies was 29.04 (SD=58.13; CI=(19.48, 37.69)). Employing the correction to account for the fact that 26.9% of the respondents are not indifferent, the mean expected WTP to increase crows and magpies in Berlin was: 0.269*29.04=7.81 USD/year (CI: (5.24, 10.14)). Overall, the distribution of expected WTP for corvids differed across the two cities, with Seattleites, on average, willing to pay (as a household) a small amount to reduce the abundance of crows, while Berliners were willing to pay a small amount to increase the abundance of crows and magpies (Fig. 1).

In Seattle, positive conservation attitude was the only significant variable impacting the WTP to fund public efforts to increase the abundance of finches (Table 3). In Berlin, residents were willing to pay more for finches if they belonged to a conservation organization but less if they lived in a population dense area (Table 4). For corvids, Seattleites belonging to a conservation organization were willing to pay more to increase crows and to a lesser extent so were those living in population dense areas. Conversely, females and residents who owned their homes were willing to pay more to reduce crows (Table 3). In Berlin, belonging to a conservation organization, fulltime employment, and being a foreign-born resident had a positive impact on willingness to pay to increase corvids (Table 4). In contrast, Berlin residents who were older or reported finding crows or magpies bothersome had a lower WTP to increase their abundance. Further, Berlin respondents with higher levels of education were willing to pay less (with the impact of age and education less than additive, as evidenced by the positive age-education interaction).

Aggregated values

In terms of aggregate WTP, and using the same household population statistics as for the expenditure model, we estimate that for Berlin, the total WTP to fund public programs aimed

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[1] This kind of correction implicitly assumes that the respondent's decision to answer "yes" to the WTP conditioning question is independent of observed covariates. We tested this assumption by estimating a binary response (logit) with a range of covariates and were not able to find significant predictors of this choice.
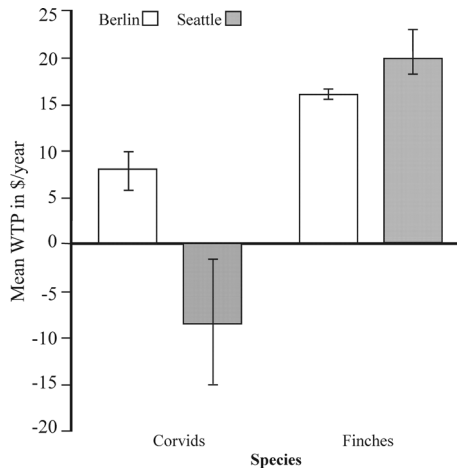
Fig. 1 Mean willingness to pay (*WTP*) for corvids and finches in Berlin and Seattle in US dollars with 95% confidence intervals derived by the Krinsky-Robb procedure (see text for details)

at conserving songbirds is: 15.80*1.99=31.4 million USD/year (95% CI: (30.33, 32.3)) to increase abundance of finches and an additional 7.81*1.99=15.5 million USD/year (95% CI: (10.4, 20.2)) to increase abundance of corvids. Adding in the revealed preference (recalled expenditure) estimate of 23.4 million, the point estimate for total economic value of common urban songbirds in Berlin is at least 70 million USD/year. For Seattle, the total WTP to increase finch populations is 19.96 *1.48= 29.54million USD/year (95% CI: (26.7, 34.8)). The total WTP to decrease corvid population is 8.43*1.48=16.69 million USD/year (95% CI: (0.8, 24.2)). Adding food expenditure estimate (72.30*1.48=107.00) to the total WTP for increasing finches, and subtracting the WTP to reduce crows (meaning, at the margin, Seattle residents suffer costs from crows), we place a cautious estimate of the total economic value of common urban songbirds in Seattle to be at least 107+29.54−16.69=119.9 million USD/year.

Table 3 Seattle WTP model estimates for finches (*n* = 166; log likelihood=−167.16) and crows (*n* = 136; log likelihood =−142.89). Note: in terms of interpreting the coefficients, negative coefficients are interpreted as having a negative impact on WTP

Finches				Crows		
Parameter	Estimate	Standard Error	Pr>ChiSq	Estimate	Standard Error	Pr>ChiSq
Intercept	−3.91	11.06	0.7237	8.35	22.11	0.7055
Gender = Female	-	-	-	−33.67	17.98	0.0611
Population density	-	-	-	0.38	0.18	0.0333
Positive conservation attitude	36.89	12.67	0.0036	-	-	-
Conservation organization membership	-	-	-	52.09	22.61	0.0213
Own home	-	-	-	−33.95	21.08	0.1072
(error standard deviation)	64.69	6.75		83.08	8.74	

Table 4 Berlin WTP model estimates for finches (*n* = 83; log likelihood = −97.10) and corvids (*n* = 112; log likelihood = −187.35). Note: in terms of interpreting the coefficients, negative coefficients are interpreted as having a negative impact on WTP

	Finches			Corvids		
Parameter	Estimate	Standard Error	Pr > ChiSq	Estimate	Standard Error	Pr > ChiSq
Intercept	90.91	7.77	<0.0001	533.23	211.01	0.0115
Bothered by birds	-	-	-	−142.79	31.20	<0.0001
Notice birds weekly	-	-	-	−121.02	77.97	0.1206
Age	-	-	-	−7.32	2.68	0.0064
Population density	−0.09	0.04	0.0239	-	-	-
Employed fulltime	-	-	-	50.76	23.69	0.0322
Positive conservation attitude	-	-	-	-	-	-
Conservation organization membership	64.42	26.21	0.0140	69.68	39.39	0.0769
Years of education	-	-	-	−33.30	13.83	0.0161
Age* Education interaction	-	-	-	0.61	0.22	0.0059
Foreign-born	-	-	-	141.15	75.88	0.0629
(error standard deviation)	42.86	4.20	-	90.13	7.49	0.0769

Discussion

We estimate that the lower bound of the total economic value Seattleites and Berliners place on enjoyment of common urban songbirds to be 119.9 million USD/year and 70.3 million USD/year, respectively. These estimates suggest that as a whole, the cities of Seattle and Berlin place a sizeable value on the service of bird enjoyment, somewhat more so in Seattle. When we separate and compare the values of private revealed expenditures versus public stated expenditures, we find Seattleites, on average, are predicted to spend a substantially greater amount on attracting birds to their homes (72.30 USD) compared to the amount they would be willing to spend to conserve a native songbird (house finch, 19.96 USD). This difference between private and public expenditure was small in Berlin (private: 11.75 USD, public: greenfinch: 15.80 USD and corvids: 7.81 USD). We also found that Berliners and Seattleites participate in similar numbers of bird-supporting activities. Taken together, these results suggest that residents in both cities value the presence of common birds around their homes.

Nationally, birds are the most observed wildlife in the United States and about 50 million people have been reported to feed birds (20.9% of the adult population) spending a total of approximately 4 billion dollars (U.S. Department of the Interior et al. 2011). Our estimated mean expenditure on bird food in Seattle is greater than the national estimate of about 17 USD per adult per year in the United States, which could be due to the fact that the state of Washington has one of the highest percentages of bird activity participation (31% compared to a national average of 21%; Carver 2009.

In both Seattle and Berlin, WTP for finches was greater than for corvids. On average Seattleites actually stated they were willing to pay to reduce American crows, however, most respondents appeared indifferent about crows (notably, male respondents have a WTP not significantly different from zero). Thus, we find that public measures aimed at reducing crow populations would be controversial given the large disparity in preferences across Seattleites, particularly since members of conservation organizations have a positive WTP to increase

crows. In contrast, Berliners on average were willing to pay a small amount to increase corvid abundance. This difference in WTP for corvids between cities, given that both tend to have a slightly negative view of these species (Clucas et al. 2011), may be due to cultural differences (e.g., Kellert 1994). Previous surveys on attitudes towards wildlife suggest that Germans generally have "moralistic" (concern for the mistreatment of animals) and "naturalistic" (affection for wildlife and nature) views of wildlife, which is in contrast to the more "utilitarian" (interest in practical value of animals) and "negativistic" (indifference, dislike or fear of animals) views found in the United States (Kellert 1994).

In comparison to other WTP studies on birds, the average WTP for finches in Seattle is less than what Americans were willing to spend on endangered species nationally (e.g., 37.56 USD for bald eagles and 76.58 USD for northern spotted owls in Stevens et al. 1991 and Rubin et al. 1991 as reviewed in Kaval and Roskruge 2009) or common game species (e.g., 23.10 USD for wild turkey; Stevens et al. 1991). These endangered and game species are also larger than finches, a factor known to influence the attractiveness of birds (Clucas et al. 2008). It may also be that rarely seen species are given a higher value due simply to their rarity, with the marginal value for common birds being lower due to their relative abundance (see Booth et al. 2011).

Different factors influenced bird food expenditure, non-monetized bird-supporting activites, and stated WTP (Table 5). Not all factors we expected to play an important role were found to be significant. For example, income was not generally found to be an important factor in our models, although in Berlin, residents with greater income levels spent more on bird food and participated in more bird-supporting activities. As expected, however we found that older residents in both cities spent more on bird food and were either more likely to try to attract birds (Berlin) or engage in more bird activities (Seattle). This is similar to a recent study in England (Davies et al. 2012), further suggesting that urban birds represent a significant opportunity for wildlife interactions among older residents. We also found a fairly consistent relationship that pro-conservation attitudes and membership in conservation organizations led to higher levels of revealed and stated preference for birds in both cities. Population density and home ownership had mostly negative and positive effects, respectively, which may stem from the fact that residents living in denser areas have less access to space to put up bird feeders while home owners are more likely to put up permanent feeders than renters (but see Davies et al. 2012). Educational attainment had a positive impact on bird food expenditures in Seattle, but a slight negative one in Berlin (although higher educational attainment makes it more likely for Berlin residents to engage in bird-supporting activities).

Overall, the values presented need to be interpreted cautiously. First, our analyses focus on mean expected expenditures and expected WTP levels, whereas the in-sample heterogeneity and individual-level preference heterogeneity was sizeable. Second, the expenditure estimates are based on recall, which introduces additional noise in the data as some respondents could not estimate how much they spent on bird food. Third, one should keep in mind that the bird food expenditures (and non-monetized activities) apply to all urban bird species residents of Berlin and Seattle may find appealing, while our stated preference WTP questions deal only with finches and corvids. Fourth, several variables expected to be significant were not. Finally, one obvious source of potential bias is the presence of residents at home at the time of the survey. Although some surveys were conducted outside of normal business hours, such a sampling strategy is likely to over-represent those who do not work outside of home. However, we collected information on age and employment status of the respondents and used those variables in our models and we used Census information for aggregation. Thus, unless the selection of the sample is based on unobservable respondent characteristics, we do not believe our results were biased as a result.

Table 5 Qualitative impact ("+" = positive, "-" = negative) of different variables considered across models and cities (cells without a symbol were not significant)

Variable	Expenditure model		Activity model				Stated preference WTP model (corvids)		Stated preference WTP model (finches)	
	Seattle	Berlin	Seattle		Berlin		Seattle	Berlin	Seattle	Berlin
			Participation	Level	Participation	Level				
Bothered by birds					-	+		-		
Notice birds weekly		+								
Gender = Female		+		+	+		-	-		
Age	+	+		+	+		+	-		-
Population density	-	-	-		-		+			
Employed full time			-	+				+		
Positive conservation attitude	+	+	+		+	+	+	+		
Membership in an environmental organization		+			+	+	+	+		+
Own home			+		+	+				
Years of education	+				+	-		-		
Age*Education interaction	-				-	+		+		
Foreign-born								+		
Income category		+				+				

On the whole, our results suggest that a sizeable economic value is derived by urban populations from the presence of common, native, "backyard" songbirds. Furthermore, these birds can provide several indirect ecological services for humans such as seed dispersal (e.g., Hougner et al. 2006) and reducing insect pests (e.g., Heyman and Gunnarsson 2011), which if the value were to be estimated, would increase our estimates.

Clearly, concerns associated with using stated preference (contingent valuation) methods to place monetary values on goods of nature such as urban birds apply to our work. These concerns range from ethical objections to economic valuation of nature (e.g., Sagoff 2011) to some neoclassical economists' objections about the economic validity of stated preference estimates (Hausman 2012). Salles (2011) provides one nuanced view of economic valuation, suggesting that "the choice is not between valuing or not valuing, it is between valuing with explicit and contestable methods and valuing implicitly". In terms of economic validity of estimates, we point to views of Kling et al. (2012) and Haab et al. (2013) and acknowledge that while stated preference estimates may suffer from known biases, we made an effort to minimize those and our estimates can play a useful (although by no means dogmatic) role in urban ecosystems management.

Most valuation studies related to birds have focused either on threatened or endangered species or on value of hunting or travel to bird-watching sites (e.g., Loomis and White 1996, Richardson and Loomis 2009, Edwards et al. 2011, Knoche and Lupi 2013, Czajkowski et al 2014), but when considering the potential benefits birds provide humans on a daily basis, it is also the common, backyard, species that require attention and generate significant economic value. Bird diversity in urban areas has the potential to increase human well-being, as well as economic value of an area (e.g., Farmer et al. 2011). Bird diversity is also related to human demographic and socio-economic characteristics in urban areas (Kinzig et al. 2005; Strohbach et al 2009). In general, areas of greater socio-economic status have higher bird diversity (reviewed in Strohbach et al. 2009), which has the potential to create an ecological inequality. Thus, restoration and maintenance of green spaces that increase bird habitat and diversity in urban areas has the potential to benefit humans in deprived areas, providing the valued service of bird enjoyment. Our estimates suggest that the economic value of such public investments in urban birds is likely to be sizeable in both the United States and Germany.

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