Effect of Urbanization on Biodiversity:
Sociological and Ecological Perspectives

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ABSTRACT

One of the most pressing issues facing not only the United States but the world is the accelerated loss of biodiversity. The leading cause of declining biodiversity, from an ecological point of view, is the destruction and fragmentation of habitats; urbanization is almost second to none in this regard. The literature is rich with studies showing the detrimental impacts of increasing levels of urbanization on biodiversity. The bulk of these studies are ecological spatial analyses, mainly focusing on avifauna. Sociological literature is not overly saturated with research relating to biodiversity loss, particularly on the sub national level of the United States, where such research is none existent. Considering this, we attempted to analyze the relationship between increasing urbanization and bird biodiversity via bivariate correlations and linear regression models. Here, we test different sociological/ecological theories, namely: Urban-Rural Dynamics (URD), IPAT, and Ecological Kuznets Curves (EKC). We mainly focused on URD but saw this as an opportunity to test other relevant theories for which we had sufficient data. Support was found for URD and partial support was found for the IPAT model however, our results for EKC were inconclusive. Research was limited by the data available, which was expected, as this is the first study of its kind to be done on the sub-national level of the United States.

Introduction: Trends in Biodiversity Loss and Urbanization

It is theorized that that there are three main measures of environmental degradation, known as the three planetary boundaries; disruption of the nitrogen and carbon cycles being two of them, biodiversity loss being the third (Clement, 2016). Currently around 1.6-1.7 million types of eukaryotes (domain which encompasses animals, plants, and fungi) have been identified worldwide with about 15,000 added each year (May, 2011). Unfortunately, we are seeing a large number go extinct each year, and more worrisome is the extinction of populations that are the functional units relative to ecosystem functioning (Ceballos, 2015). The background extinction rate (rate at which extinctions tend to happen in-between mass extinctions) is around 2E/MSY (that is 2 extinctions per million species per year), currently it is approximated that the actual extinction rate is around 8-100E/MSY and this estimate is considered conservative (Ceballos, 2015). Furthermore, we are not sure exactly how many species are on this planet, and of the ones we do know of, only about 5% have been evaluated for their behavior ecology (how they interact with their respective ecosystems) (May, 2011).

Threats to biodiversity, threaten ecosystem functioning and in turn threaten ecosystem services; the benefits we receive from the environment, which we all depend on. Services such as: carbon storage, carbon sequestration, water purification, food production, forest and medicinal products, recreational activities, nutrient cycling, soil erosion prevention and many more (Costanza, 1997; TEEB). The effects of biodiversity on ecosystem functioning (underpinnings of ecosystem services) is well established in the scientific community as it can impact processes such as: pollination, seed dispersal, regulation of climate, pest control, etc. (Diaz et al., 2006). For example, it has been shown that increased species richness (total number of species found in an area, an indicator of biodiversity) across trophic groups promotes high threshold functioning of multiple ecosystem functions. Additionally, other research shows that having multiple types of tree species enhanced ecosystem services delivered in a large production forest region (Lefcheck et al., 2014, Gamfeldt et al., 2012). Furthermore, single species were not able to sustain the same amount of functioning and/or benefits to a comparable level (Lefcheck et al., 2014, Gamfeldt et al., 2012).

One of the driving forces of biodiversity loss is land use intensification, specifically those associated with urbanization, coupled with the ever-increasing agricultural intensification needed to support cities and growing populations (Fulkerson and Thomas, 2014; Czech et al., 2000; Clement, 2016). The world became predominately urban (greater than 50%) in 2008, and it is estimated that by 2050 66 % of the global population will live in some form of urban environment (UN, 2014). Globally, somewhere between .5% and 3% of the world’s surface is utilized as urban spaces. Despite being such small areas urban centers cannot exist without intensive resource importation from peripheral rural (e.g. crop and pasture lands) lands which encompass approx. 10% and 25% of the earth’s surface respectively, further increasing the potential impact urbanization will have on biodiversity (Fulkerson and Thomas, 2014; Clement, 2016). At the finer scale of the United States we see similar trends. Around 80 % of the population resides in urban areas, and it was estimated in 2007 that about 3 % of the land use in the United States is urban while cropland and pasture land take up 18% and 27% respectively (UN, 2014; Nickerson et al.,2011). About 60% of the United States’ imperiled, rare species are found in areas designated as metropolitan and in a study done on the 35 fastest growing metropolitan areas, it was found that 29% of these imperiled species are found within them (Ewing et al.2005). From 1945 to 2007 urban land use, by acreage, was increased by a factor of four, with the pace of land development accelerating each decade, putting imperiled species at increased risk (Ewing et al., 2005; Nickerson et al., 2011).

Literature Review

Ecological Research and Theory: Urban Effects on Biodiversity

While we see that trends of urbanization are increasing both globally and within the United States (UN, 2014) ecological research into the effect of urbanization on biodiversity show that it is a mostly a negative relationship. Research is primarily concerned with the overall loss of biodiversity, usually indicated by the species richness metric and Shannon diversity indexes (Blair, 1996; Blair, 2004; McKinney, 2002; McKinney, 2008; Chase and Walsh, 2004; Van Nuland and Whitlow 2014; Arson et al.2014); as well as the homogenizing effect urbanization has on populations, utilizing Jaccard's index and evenness measurements (Blair, 2001; McKinney, 2005; Shochat et al.2010). Studies typically organize the study area along a "rural to urban gradient." The purpose of the gradient is to compare changes in whichever metric of biodiversity is being measured, as you move towards an urban core, to see how an increase in built surfaces effects biodiversity (McKinney, 2002). Some studies have also considered the temporal effects of urbanization on biodiversity (Tait et al.2005; Van Nuland and Whitlow 2014) but they are less common, most likely due to lack of records (McKinney et al., 2005). These ecological studies on urbanization and biodiversity, have found that while yes, urbanization generally has a negative impact on biodiversity levels; the relationship is not always linear. Many studies mention the Intermediate Disturbance Hypothesis (IDH) which simply postulates that areas of intermediate disturbance, which often have a more heterogeneous environment, will harbor the highest levels of biodiversity (Roxburgh et al., 2014; McKinney, 2002). This is not to say however, that urbanization has positive effects on biodiversity, just that those who are planning development in an area should try and increase heterogeneity (Marzluff, 2005). While we do not test the IDH in this paper, the theory should be of interest to future environmental sociology research and relates to the ideas discussed in the upcoming sections.

Sociological Research and Theory: Anthropogenic Impacts on Biodiversity

Research specifically targeted at the effects of urbanization on biodiversity is not as prevalent in the sociological literature and most studies are cross-national with higher focus on other variables (McKinney et al., 2009; Hoffman, 2004). No sociological studies have been conducted at the sub-national level, and this is no surprise as many environmental sociological theories are founded in a global context (e.g. world systems theory) or harder to test sub-nationally due to data constraints (e.g. theories of production/consumption, unequal exchange). Here we attempted to analyze two sociological theories; the IPAT model and Ecological Modernization theory, as well as the associated Environmental Kuznets Curve.

The IPAT model is theoretical equation which predicts that environmental impacts (I) are equal to population (P) x Affluence (A) x Technology (T) (Ehrlich and Holdren 1971). It was given a stochastic overhaul sometime later (see Dietz and Rosa 1994) to allow for hypothesis to be tested statistically; this reformulation was called STRIPAT. The STRIPAT model adds elasticity to population and affluence variables of the models while calculating for error inherent due to variables not being included in the model (e.g. technology) (Dietz et al., 2007). McKinney et al., (2009) found support for the IPAT model during a cross-national analysis on bird biodiversity loss and noted population as having a more pronounced effect than affluence. Similarly, Dietz et al., (2007) also shows strong support for the IPAT model, while additionally concluding that other variables such as urbanization have little effect on anthropogenic environmental impacts. It should be noted however, that Dietz et al., 2007 used ecological footprints as their independent variable which does not account for local impacts such as biodiversity loss. As such this analysis, can help to fill in some of these gaps noted in prior sociological research concerning IPAT models.

Ecological Modernization Theory and the related Environmental Kuznets Curve contradict IPAT model (Dietz et al., 2007). While the IPAT model postulates that environmental impacts increase linearly with population and affluence, EMT and EKC postulate that a curvilinear relationship is present between environmental impacts and drivers such as economic development and state environmentalism (Dietz et al., 2007; McKinney et al., 2009). EMT postulates that industrialization, economic development, technological advancement and capitalism are related to environmental sustainability; Once a nation reaches a certain level of modernization, environmental reforms will begin to take place (York and Rosa, 2003). EKC are often used in studies associated with EMT and they show a non-linear relationship between a local environmental impact (such as biodiversity loss) and increasing economic development (usually per capita GDP); the idea is that initially economic growth is going to cause environmental impacts but a point is reached where increasing economic growth begins to decrease impacts (York and Rosa, 2003; McKinney, 2009). As such, these theories are also related to Maslow’s hierarchy of needs as they work off the assumption, that as a country meets some level of economic need, it will begin to assess other needs such as ecological sustainability. However, EMT and EKC research has multiple flaws as pointed out by York and Rosa, (2003). One of the biggest being that studies are usually done locally and cannot account for the well-known “Netherlands Fallacy” meaning they cannot account for environmental impacts that moved outside the borders of a nation. This means that a nation can in-fact seem to have an EKC type relationship between development and environmental reform; however, impacts have just been moved out to other peripheral or semi-peripheral nations (as outlined in world systems theory) (York and Rosa, 2003; McKinney, 2009). One such study specifically focused on biodiversity EKC, Mills and Waite, (2009), did not find support for EMT/EKC despite using improved statistical analysis (quantile regressions and spatial filtering). However, they also note that more research is needed into the relationship between affluence and biodiversity, concluding that policy should not be informed by any research involving EKC, whether the results be a positive or negative find.

Methods

Dependent Variable and Hypothesis

In this paper, our main objective was to investigate the relationship between biodiversity and urbanization at the sub-national level of the United States. We also saw this as an opportunity to investigate some sociological theories that try and predict environmental impacts (e.g. IPAT and EMT/EKC). To our knowledge this study is the first of its kind. We began by searching for a dataset which contained biodiversity data for every state, and we found it via the Nature Conservancy's 2002 State of the Union: Ranking America's Biodiversity (See Table 1). This is the only such dataset that accounts for every state, to our knowledge; containing data for six taxonomic groups: Birds, Amphibians, Reptiles, Fresh water fish, Mammals and Vascular plants. We decided to use bird biodiversity (specifically % at risk) as our dependent variable in this study due not only to the fact that much ecological and sociological research has also focused on birds; birds have also been shown to act as surrogates of other species (Blair, 1999) and meet several criteria (not all) of the attributes needed for a species to be considered an indicator species (Blair 1999; Noss, 1990). Additionally, we ran bivariate correlations between all taxonomic groups and founds birds to correlate highly with almost every group excluding mammals and reptiles (See Table 2). We hypothesized that increasing urbanization would have a negative impact on the % of birds at risk and account for the most impact of all models tested. We also expected to see evidence for the IPAT model, meaning affluence (for which technology is also implied) and population would negatively affect biodiversity as they increased. Lastly, to test EMT we investigated the possibility of an EKC for % birds at risk, we did not expect to see the EKC curvilinear relationship, believing we would likely see a linear relationship between birds at risk and increasing affluence, as predicted by IPAT and URD expectations.

Independent Variables:

Urban Rural Dynamics

As Stated above, our focus was on the relationship between urbanization and biodiversity loss. We used change in urban land cover by state from 1987 to 1997 as well as change in cropland by state (to represent rural areas) (See Table 1). For every independent variable, we had some measure of lag time, around 3 to 5 years, as the biodiversity data set was not produced until 2002.

IPAT

We tested the IPAT model by finding datasets for both affluence and population, as studies have in the past (McKinney et al., 2009). For population, we used data from the U.S Census bureau for the year 2000 by state, for affluence we use a median household income set of data that contained each state for the year 2000; this differs slightly from past studies such as McKinney et al., (2009) and Dietz et al., (2007) which use per capita GDP, this metric is typical in sociological studies (See Table 1 for all data sources).

EMT/EKC

We could not specifically test EMT as we had no measure of state environmentalism which is needed to truly test the theory (York and Rosa, 2003; McKinney et al., 2009). Here we specifically test for the EKC between % birds at risk and affluence (See Table 1). While EKC is not the same as EMT, they postulate similar ideas, mainly that there is a non-linear relationship between environmental impacts and economic development, and late state development within nations see environmental improvement (Dietz et al., 2007; Mills and Waite, 2009).

Sampling and Analysis Technique

Our sample size contained all states of the U.S. as well as the District of Columbia (n=51); as such we had data representative of the true population mean for the U.S. We found our main dataset for biodiversity to skewed so we implemented a natural log transformation for all variables being tested in SPSS (all statistical analysis was conducted in SPSS). We then ran bivariate correlations between our dependent variable and all independent variables to have a starting place for our later analysis in which we generated models using linear regressions in SPSS (See Table 4). Models generated via SPSS were then interpreted in the context of the above-mentioned theories to be tested; our cut off for significance was .10, given the small number of cases (n=51) and considering that this was a total population, rather than a sample.

Table 1: Main variables used and their sources (See Reference Page for More Info)

Variable	Source
% Birds at Risk by State	The Nature Conservancy 2002
Land Use Data (Urban and Crop land)	USDA Major Land Uses
Population by State 2000	U.S Census Bureau
Median Household Income by State 2000	U.S Census Bureau

Table 2: Bivariate Correlations for taxonomic groups and % birds at Risk 2002.

Taxonomic Group	Correlation with % Birds at Risk 2002
% Mammals at Risk 2002	
- **Pearson Correlation**	- .267
- **Sig. (2 Tailed)**	- .058
% Reptiles at Risk 2002	
- **Pearson Correlation**	- .231
- **Sig. (2 Tailed)**	- .103
% Fresh Water Fish at Risk 2002	
- **Pearson Correlation**	- .549 (***)
- **Sig. (2 Tailed)**	- .000
% Amphibians at Risk 2002	
- **Pearson Correlation**	- .411 (**)
- **Sig. (2 Tailed)**	- .003
% Vascular plants at Risk 2002	
- **Pearson Correlation**	- .601 (**)
- **Sig. (2 Tailed)**	- .000
% All Species at Risk 2002	
- **Pearson Correlation**	- .675 (**)
- **Sig. (2 Tailed)**	- .000

†. Correlation is significant at the .10 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed).**. Correlation is significant at the 0.01 level (2-tailed) ***. Correlation is significant at the .001 level (2-tailed)

Analysis

Univariate & Bivariate Analysis

In table 3 (below) you will find descriptive statistics for all variables in this analysis. As stated above our dependent variable (% Birds at Risk) was a non-normal distribution and so we have natural-log transformed all our variables to meet the assumptions of linear regressions (McKinney et al., 2009). A Bivariate analysis was run in addition to the univariate analysis, as to see the correlations between our dependent variable and the independent variables (Table 4). Change in urban area and population by state had the only strong, significant correlations (see Table 4) with our dependent variable; showing that urbanization and increased population likely have a negative impact on bird biodiversity. None of the other main variables tested had a significant correlation with our dependent variable, on a bivariate level.

Table 3: Descriptive Statistic for all Variables analyzed (natural logged)

Variable	N	Mean	Median	Mode	Std. Deviation
% Birds at Risk 2002	51	.6251	.5878	.59	.63849
Urban Area in acres (87'-97')	25	.2247	.1413	-.09	.30059
Crop Area in acres (87'-97')	12	-.1431	-.1786	-.30	.12018
Population 2000	48	15.0915	15.2433	13.11	1.02975
Median Household Income 2000	50	10.6281	10.6311	10.29	.15716

Table 4: Bivariate Correlations Between the Independent Variables and Dependent Variable

Independent Variable	Correlation with % Birds at Risk 2002
Change in Urban Area (87'-97) - **Pearson Correlation** - **Sig. (2 Tailed)**	 - .589 (**) - .002
Change in Crop Area (87'-97') - **Pearson Correlation** - **Sig. (2 Tailed)**	 - .139 - .668
Median Household Income by State 2000 - **Pearson Correlation** - **Sig. (2 Tailed)**	 - .096 - .506
Population by State 2000 - **Pearson Correlation** - **Sig. (2 Tailed)**	 - .536(***) - .000

†. Correlation is significant at the .10 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed). **. Correlation is significant at the 0.01 level (2-tailed) ***. Correlation is significant at the .001 level (2-tailed)

We began our multivariate analysis by first performing OLS regressions for our two variables with significant correlations in our prior bivariate analysis, urban land area changes and population by state (Table 5; Models 1 and 2 respectively). Model 1 had explained approx. 32% of the variation in our dependent variable while Model 2 explained 27% however; Model 2 had a slightly higher significance level and a larger F-Score (See Table 5). It seems that both increasing urbanization and population at the state level are relatively equal in their impacts on biodiversity. Seeing as both variables seem to be good predictors of biodiversity loss, a model was created to see how the two interacted together with the dependent variable (% bird species at risk). Interestingly, this model called Model 3, shows urbanization as non-significant while population stays significant and has a higher standardized beta coefficient; a perplexing result which we discuss in the next section. We then further tested our URD theory and IPAT model in the Models 4 and 5. Model 4 test the urban variable with the change in crop land area variable, this model just barley meets our cut off for significance (p-value=.099) but yields interesting results. Here only the urban variable is significant however; this model explains the most variation (approx. 34%) in our dependent variable of any model, the standardized beta coefficient for our urban variable also increased but had a lower significance. Model 5 is our STRIPAT model and explained about 33% of the variation in our dependent variable. Both the population and affluence variables were highly significant however; affluence had an inverse relationship (e.g. negative beta coefficient) which is contradictory to the IPAT theory, yielding mixed support. Lastly, Model 6 tested for an EKC for % birds at risk by state. Here we used our affluence variable and the square of this variable to test for a non-linear U-shaped type relationship with our dependent variable. Initial tests were not significant until we added in our population variable. While this model is highly significant, explains a decent amount of variation (approx. 33%) and yielded a non-linear, negative relationship between the square of median household income and our dependent variable; the model had a high degree of multicollinearity and the initial affluence variable was excluded. We do not feel that interpretation of this model is viable and our results for EKC are inconclusive.

Table 5: Regression Models and their corresponding F-stats, variance, beta coefficients and significance.

Stat	Model 1	Model 2	Model 3	Model 4	Model 5	Model 6
F Score	12.223** .002	18.555***	4.709*	3.275†	12.273***	12.291***
Adjusted R^2	.319	.272	.261	.336	.329	.329
Urban Land Area Change (87'-97')	.589*** .002	X	.140 .482	.699* .041	X	X
Crop land Area Change (87'-97')	X	X	X	-.015 .958	X	X
Population by State 2000	X	.536*** .000	.599** .006	X	.583*** .000	.583*** .000
Median Household Income by State 2000	X	X	X	X	-.344** .010	Excluded
Median Household income 2000 (Squared)	X	X	X	X	X	-.335** .010

†. Correlation is significant at the .10 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed).**. Correlation is significant at the 0.01 level (2-tailed) ***. Correlation is significant at the .001 level (2-tailed)

Discussion and Conclusions

Bivariate Analysis

Our bivariate analysis yielded some interesting results. Urban land area change had the highest correlation and second highest significance. This was expected as numerous studies have shown such an effect, from local scale ecological studies to large scale cross national sociological studies as was stated in the literature review. While our results are preliminary and the relationship between urbanization and biodiversity loss is complex and likely encompasses far more variables then just the few used here, the fact that such a relationship likely exists at most spatial scales contemplated is cause for concern. Population was our other significant predictor of increased risk to biodiversity loss, meaning higher population levels likely results in higher local environmental impacts. Our affluence variable had no significant effect which is not in line fully with the IPAT model. Lastly our other variable, change in crop land area also had no significant effect. This result reflects the effect of agriculture on biodiversity; depending on variables such as the method of cultivation, fragmentation of habitat patches and intensification of the practice, agriculture can have both positive, negative and null effects on biodiversity levels.

Multivariate Analysis

Our multivariate analysis worked off the findings in our bivariate analysis and yielded results that were mostly predicted for the theories tested. As expected, both the models including our population and urban variables were significant and showed negative impacts on the % birds at risk by state. When the two variables were used together however, an unexpected result occurred; population stayed significant while change in urban land area was insignificant. It appears that some unexplained interaction effect occurs between population and urban land area change in which the latter effect is no longer significant. We can provide no explanation for this effect but speculate that the low temporal resolution of our dependent variable may be a contributing factor. Additionally, a county level analysis would be preferable, were such data available.

Models 1 and 4 provide support URD theory that increasing urbanization negatively impacts biodiversity. Change in urban land area was the best sole predictor while when combined crop land are changed, contributed more to the model but had a lower significance. The implication of this is that these models further the support for URD and suggest an interaction between crop area and urban areas. Agriculture is part of URD theory as urban centers require significant contributions from peripheral rural crop and pasture land area to exist. Cropland increasing the effect of urban land area change, seems to provide support for these ideas. Because cropland itself is not significant, along with its interaction effect on urban land area change, we cannot conclusively say the impact of this variable.

Our multivariate analysis also shows partial support for the IPAT model. Population was a significant predictor in all models it was involved in though results for affluence were contradictory to the IPAT model. Model 5 explicitly tested this and showed population to be very significant predictor of increased to risk to bird biodiversity though affluence had an opposite but significant effect. The fact that affluence seems to reduce the risk to bird biodiversity was unexpected indeed, and seems to be more in line with the theories of EMT or EKC. We are not sure why such an interaction occurred but seeing as our sub-national analysis likely falls victim to the “Netherlands fallacy” (York et al., 2003) it may be due to the absence of data relating to the unequal exchange between the United States and peripheral nations (McKinney et al., 2009;

Shandra et al., 2009). Put simply, we may be exporting the problem through displacement, and not truly solving it at its source.

Lastly our test for an EKC between affluence and % birds at risks was inconclusive though the model suggests that some type of non-linear relationship may be present. The high multicollinearity of this model makes it inconclusive along with the exclusion of regular affluence variable. We cannot give a proper explanation for this model now, and do not suggest interpreting the model, however, it could be a possible baseline for future analysis. Residualization may offer one viable way to resolve the multicollinearity problem

Future Research and Conclusions

Our analysis was the first of its kind at the sub-national level of the United States at the state level. We attempted to test three main theories: URD, IPAT and EKC, mainly focusing on the URD. We found significant support for URD, partial support for IPAT, and EKC results were inconclusive. This study was limited mainly by the low temporal resolution cross section of our dependent variable data set, for which only one year was represented and the methods of data collection for the dataset were not definitively stated within the source. Additionally, being the first study at this scale, was another limiting factor as we did not have much prior research to work with. The most asset gained from our analysis may very well being implications for future research directions. While we found strong support for URD interaction effects with population and cropland are currently unexplained, furthermore the relationship is likely more complex and involves more variables than just the ones tested in this study. Similarly, we found partial support for IPAT but interaction effects between populations and other variables go unexplained, while the effect of affluence in the IPAT model contradicts the main theory and may provide some support for EKC. However, our EKC model was inconclusive as such we have nothing reliable to report in relation to this theory. As such, future research should first focus on attempting to expand the database for nationwide biodiversity metrics as seen in the Nature Conservancy's 2002 report; higher temporal resolution (longitudinal analysis) would be of great assistance in the researching of the different effects land uses such as urbanization have on biodiversity. Though we provide support for URD, an article by McDonald et al., (2013) conveys an important point: urban areas are not going anywhere and should be treated as an inevitability for which we need to find solutions, not explicitly a problem or a solution. Considering this point, future research into the relationship between biodiversity and urbanization should try and focus on testing solutions such as urban corridors, green space and canopy cover (Austin, 2012). Promoting sustainable development within urban centers can help to lessen their impact, and research into these areas can help make such development occur. If we wish to conserve biodiversity and the ecosystem functioning and services that high levels of biodiversity support, we must be realistic and practical when examining complex relationships such as the one between urbanization and biodiversity loss.

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