Urban Heat Island Effects on Moth Phenology

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ABSTRACT:

Because only large-scale studies have shown global climate change and land-use conversion altering life cycle timings among species, there has not been enough attention towards analyzing effects of temperature on phenology over smaller time scales. Some of these alterations include earlier flowering in plants and earlier migrations in birds. In this study, we looked at how moth phenology has been affected by the impacts of urban heat islands across an urban-intermediate-rural gradient. Using blacklight traps to capture moths, we examined the emergence of moths to determine where and when phenological timings were occurring. Results from the survival analyses showed that four out of the 12 species of moths we examined had an earlier emergence at the urban site compared to the rural site. This supports our hypothesis that emergence would occur sooner at the urban site as the urban site would be warmer than the rural. By studying the effects of urban heat islands on phenology, we can have a better understanding of how broader concerns such as global climate change and deforestation will affect organisms.

INTRODUCTION:

Global climate change is expected to increase temperatures by 1.5-4 °C by the end of the century (IPCC 2013), and species’ phenologies are expected to be altered by these rising temperatures. Among all major taxonomic groups, studies have shown that the altering phenology of species has been linked to climate changes (Parmesan 1996, Parmesan 2006). Developmental rates among numerous species have increased as a result of climate warming, causing their development to begin earlier in the season (Frochhammer et al. 1998; Post et al. 2008). As a result of these shifting phenologies, species’ distributions have been altered due to physiological constraints that make current habitats unsuitable (Parmesan 1996, Root et al. 2003). Some species have been moving to cooler habitats, such as higher elevation or outside urban centers, where it is more compatible for their survival (Grimm et al. 2008).

Phenological timing among Lepidoptera, which includes butterflies and moths, has also been altered due to changes in regional climates. One study conducted on butterflies in lowland California found that over the 31 years of recorded climate data, the average shift in flight time of species was up to 24 days earlier (Forister and Shapiro 2003). In addition, climate change is disturbing lepidopteran habitats through alterations in host plant development, therefore altering phenological events. Egg-laying rates, for
example, are temperature-dependent as warming temperatures are affecting host plant development, leading to shifts in where eggs are being laid within microhabitats (Davies et al. 2006).

While the effects of climate change on long-term phenological processes can be difficult to test experimentally, using existing temperature gradients created by anthropogenic impacts can be used as models for future climate scenarios (Carriero and Tripler 2005, Grimm et al. 2008). These gradients can forecast temporal data across spatial scales in what is known as space-for-time substitution (Blois et al. 2013). In our study, we used an urban-rural gradient to examine temporal differences as a result of both climate change and land-use change. This allows us observe biological changes in a shorter amount of time rather than waiting for the full effects of climate change.

Land-use changes, such as urbanization, have had increasing impacts on species as cities become larger both in total area and population, creating urban heat islands. Urban heat islands (UHIs) are defined as metropolitan areas that are warmer due to the greater amount of impervious surfaces than their surrounding rural areas (EPA 2014). Urbanization can increase local air temperature 1-3 °C and surface temperatures 5-9 °C, compared to rural temperatures (Imhoff et al. 2010; Karl et al. 1988; Stone 2007). Impacts of heat islands can include alterations in both plant and animal communities, as species may shift their distributions away from urban environments to escape warming temperatures (Grimm et al. 2008).

Variation in temperatures can lead to differences in biological processes such as metabolic rate and reproduction, particularly among ectotherms as they are highly dependent on temperatures within their environment. Insects, including lepidopterans, are highly abundant and diverse, making them good indicator species for determining the extent of environmental impacts (Rosenberg et al. 1986; Diamond et al. 2012). While increasing temperatures from both climate change and urbanization are thought to cause more rapid development, temperatures may reach a point where phenology becomes delayed as conditions may be outside a species’ optimum range of temperatures ideal for development (Ibáñez et al. 2010; Nufio et al. 2010; Diamond et al. 2014). In lepidopterans, some species are emerging earlier in rural sites than in urban sites as their thermal optimum is lower (Diamond et al. 2014). This concept could help explain why there are distributional differences in relation to temperature among different species of lepidopterans.

We measured the phenology of 12 focal moth species at three locations along an urban-suburban-rural gradient in order to examine urban heat island effects on moth development. We used survival analyses to test whether moths emerge earlier at urban locations than at rural locations. We predict that if rates of development increase in response to temperature, then we will see earlier emergence of moths at an urban site than at a rural site.

**METHODS:**

Moths were collected at three sites: the Birmingham-Southern College Forest (BSC, an urban site), Ruffner Mountain Nature Preserve (RMNP, an intermediate site), and Turkey Creek Nature Preserve (TCNP, a rural site). All of these sites are located in Jefferson County, Alabama, USA. Birmingham-Southern College Forest, (33.514°N, 86.858°W) has an elevation of 192 m with 6.5 ha of deciduous forest and a southeast-facing aspect. Ruffner Mountain Nature Preserve (33.534°N,
86.711°W), has an elevation of 249 m with 400 ha of deciduous forest and a southeast-facing aspect on the outskirts of Birmingham. Turkey Creek Nature Preserve (33.70°N, 86.70°W) has an elevation of 200 m with a 180 ha forest surrounded by much larger expanses of deciduous forest. This site also had a southeast-facing aspect and was located 32 km outside of the Birmingham metro area. The following sites were categorized as urban, intermediate, and rural dependent based on proximity to the Birmingham metro area and the amount of impervious surfaces in the surrounding areas.

We chose 12 focal species to monitor throughout this experiment (Table 1). Focal species were chosen based on their presence at each site, abundance, and ease of identification. Our analysis of each species was limited to one cohort of individuals to ensure that the first emergence of individuals within each species was captured during our trapping days.

We used two blacklight traps at each location to capture moths. Each trap had a light attached in the center that attracts the moths with jars of ethyl acetate in the bottom of the bucket to euthanize the moths. The traps use a 15 watt Hg bulb that is powered by a 12 volt lead-acid battery. The traps attracted moths within a 50 m radius (Summerville and Crist 2003). Traps were set twice a week in the evenings and collected the following morning, where moths were identified and labelled. In this study, we did not sample any moths that are diurnal. In addition, I used three iButton (maxim integrated, DS1921G; San Jose, CA) sensors, two placed on the limbs of trees approximately 1-2 meters off the ground and one placed in the soil (two inches in depth), at each location recording temperatures hourly over the course of the study.

Trapping began on April 25th and continued until July 4th so that species are captured during their full flight time. Once collected, moths were placed into smaller bags to be frozen so they could be pinned and sorted. Identification of the collected specimens was conducted using taxonomic keys, reference collections and recognized taxonomic experts. Because H. tessellaris and H. harrisii have indistinguishable external morphological features, identification of H. tessellaris was done by examining the genitalia of the individual as not to misidentify it.

For each of the focal species, we performed survival analyses to determine if there were significant differences in the emergence of moths at the different locations. As a non-parametric statistic, this technique allows us to compare the timing of well-defined events at multiple sites. The number of trap days was the independent variable and the cumulative percentage of moths captured was the dependent variable. This shows us when species emerged and when the numbers of individuals was greatest. However, because the total number of individuals was calculated as a proportion, it could not tell us which site has the greatest number of individuals.

In addition, we compared maximum and minimum temperatures across all sites to determine how urban heat island effects are impacting moth communities at each location. Using methods adapted from Jones et al. (2013), temperatures recorded by the iButton sensors were converted into degree-days using a base temperature of 10 °C (Jones et al. 2013). This allows us to examine the emergence of species on a time scale that is associated with temperature rather than with a calendar-day system. ANOVA was used to measure differences in heat accumulation between the three sites. We compared total degree-days across all days we recorded temperatures, as well as at different seasons to determine if there was a relationship between urban heat islands effects and seasonality. In our study, we had two replicates of degree-day totals per site, using the two sensors in the trees. Repeated measures ANOVA were used to compare maximum and minimum temperatures between the sites as well as to compare
seasonal fluctuations in temperature relative to location. All analyses were performed in SPSS (version 21.0, IBM Corp. SPSS Statistics 2012).

RESULTS:

Over the 20 collection dates, I collected several thousand moths including 6182 moths from our focal species. Some of our focal species, however, did not have at least ten individuals at one or more of the sites and therefore were not analyzed for site comparison. Of the 12 focal species, eight of them had enough individuals at each site to analyze. Out of all species, *D. olinalis* had the greatest abundance at every site out of all focal species with 2,264 total individuals across all sites.

Temperature data consisted of 5,218 observations across all three sites over a span of 218 days from December 20\textsuperscript{th}, 2013 – July 25\textsuperscript{th}, 2014. While seasonal maximum temperatures did not differ among sites, annual and summer minimum temperatures were higher at BSC than at RMNP or TCNP (annual minimum temp: F = 16.028, df = 2, p = 0.025; summer minimum temp: F = 21.366, df = 2, p = 0.017). The Post-hoc test showed that there was approximately 1°C difference in the minimum summer temperature between BSC and TCNP, with BSC being the warmer location. Degree-day calculations also indicated that there was no significant difference between sites.

Results from our survival analyses showed that four out of the eight species of moths analyzed were significantly different between the sites in terms of when they emerged (Table 2). These species that emerged earlier at one location than another were *D. olinalis*, *M. americana*, *H. tessellaris*, and *I. americanalis*. For all of these species except *I. americanalis*, there was a significant difference in emergence found between BSC and TCNP where moths collected at BSC emerged earlier (Figures 1-3). However, *I. americanalis* had a significant difference in emergence between BSC and RMNP, also emerging earlier at BSC (Figure 4). All other focal species showed no significant differences across sites.

DISCUSSION:

My temperature data supports the assumption that urban areas are generally warmer than rural areas. While there were no significant differences in overall maximum temperature or seasonal maximum temperature, the significant difference found in overall minimum air temperature confirms that Birmingham’s urban heat island effects are stronger at night. Studies have found that atmospheric temperatures have greater differences between the city and surrounding areas at night than during the day as heat from urban infrastructure slowly releases at night after heat is accumulated during the day (Roth et al. 1989; Stoll and Brazel 1992; Nichol 2005; EPA 2014). My findings also showed that minimum air temperatures were significantly different across site only during the summer. These results, however, contradict other findings that have shown that urban heat islands have a greater effect during the winter (EPA 2014). More studies need to be conducted on urban heat island effects and their effects on seasonality.

My inconclusive findings could be due to inadequate placement of temperature sensors during seasons when trees were without leaves, which left sensors susceptible to heating through incident
radiation. This could explain the discrepancy between my findings and the results of previous studies that have shown

My findings confirm that there is variation among moth species in terms of their phenological responses to urban heat island effects. However, there were significant results that showed temperature in relation to geography is one of the underlying causes in moth emergence. As the distribution of moths extended outside the urban center, emergence of moths was delayed as the loss of impervious surfaces in the city resulted in cooler temperatures. Phenological change is often associated with temperature-driven effects on organismal physiology. For example, one study found that insect physiology has a positive relationship with temperature until it reaches a thermal optimum point above which development is delayed (Regniere et al. 2012). The thermal optimum point describes the temperature best suited for the development of an organism. As shown in other taxonomic groups, particularly among ectotherms, temperature is the main driver of metabolic rates. My results suggest that for some moth species, increased temperatures are responsible for earlier phenological timings.

As a result of earlier phenological timings in moth species, it can be speculated that other ecological interactions will be impacted as well. One study conducted on golden plovers and insects examined predator-prey interactions in response to early breeding phenology as result of warmer springs. It was concluded that golden plover breeding success will be reduced as warmer temperature will cause their egg-laying to occur sooner when prey abundance has not yet emerged (Pearce-Higgins et al. 2005). Because moth phenology is advancing, they could become more abundant prey for birds that are also hatching sooner. Another major interaction that could be affected as a result of altering phenology is between moths and plants. Multiple studies have showed that plants have begun to flower earlier as a result of climate change (Neil and Wu 2006; Parmesan 2006; Franks et al. 2007). Not only does this affect moth oviposition and food availability, but it also affects when moths pollinate, as moths are important pollinators for plants. One review showed, however, that as a result of climate change, floral resources have been reduced 17-50% to pollinators, causing disruption in plant-pollinator activity (Memmott et al. 2007). This shift in plant phenology is most likely to affect moth phenology, especially as climate change is altering community interactions.

As climate change continues to alter life cycle timings in multiple species, studying its effects will become even more crucial. However, to fully analyze climate change and its effects on biological systems requires decades of data collection that is not always practical or feasible. Substituting space-for-time can be used to model future climate projections from contemporary spatial patterns (Blois et al. 2013). Space-for-time substitution can even be beneficial for studying phenological changes in relation to climate. One study found that using spatial data were able to fill historical gaps left by using temperature data to examine phenological events (Buyantuyev et al. 2012). Although no remote sensing or other spatial data was used in this study, it could have been extremely beneficial as using temperature data has various flaws. Future studies need to be done on space-for-time substitution, particularly in relation to phenological changes.

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REFERENCES


IPCC (Intergovernmental Panel on Climate Change). 2013. Climate change 2013: the physical science basis. WMO (World Meteorological Organization) and UNEP (United Nations Environment Programme).


Table 1: Focal species, with their respective family names, used in this study. Wingspan and distribution of each species is also listed.

<table>
<thead>
<tr>
<th>Species</th>
<th>Family</th>
<th>Distribution</th>
<th>Wingspan (mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Polygrammate hebraicum</em></td>
<td>Noctuidae</td>
<td>Eastern US</td>
<td>23-39</td>
</tr>
<tr>
<td><em>Dolichomia olinalis</em></td>
<td>Pyroloidea</td>
<td>Eastern California to Florida; Texas to Nova Scotia</td>
<td>16-24</td>
</tr>
<tr>
<td><em>Malacosoma americana</em></td>
<td>Lasiocampoidae</td>
<td>Eastern US to Central US; north as Canada</td>
<td>22-44</td>
</tr>
<tr>
<td><em>Nadata gibbosa</em></td>
<td>Notodontidae</td>
<td>Throughout N. Amer</td>
<td>38-59</td>
</tr>
<tr>
<td><em>Halysidota tessellaris</em></td>
<td>Arctiidae</td>
<td>Southern Canada; western US; southern US</td>
<td>35-45</td>
</tr>
<tr>
<td><em>Spilosoma congrua</em></td>
<td>Erebidae</td>
<td>Southwestern US to southern Canada; east to Atlantic.</td>
<td>27-47</td>
</tr>
<tr>
<td><em>Chytonix palliatricula</em></td>
<td>Noctuidae</td>
<td>Eastern N. America</td>
<td>28-33</td>
</tr>
<tr>
<td><em>Homorthodes lindseyi</em></td>
<td>Noctuidae</td>
<td>Northeast from New Jersey to southwest Oklahoma</td>
<td>~ 27</td>
</tr>
<tr>
<td><em>Idia americalis</em></td>
<td>Erebidae</td>
<td>Throughout N. Amer</td>
<td>20-30</td>
</tr>
<tr>
<td><em>Atteva aurea</em></td>
<td>Noctuidae</td>
<td>Southern US to West Indies</td>
<td>~ 22</td>
</tr>
<tr>
<td><em>Marimatha nigrofimbria</em></td>
<td>Noctuidae</td>
<td>Eastern US to Illinois; southward Texas and Florida</td>
<td>18-22</td>
</tr>
<tr>
<td><em>Hypoprepia fucosa</em></td>
<td>Erebidae</td>
<td>United States and southern Canada east of the Rockies</td>
<td>25-35</td>
</tr>
</tbody>
</table>
Table 2: Focal species total abundance and p-value result of site difference in emergence. Species with less than 10 individual at one location were not included for analysis. ^ Significant difference between BSC and RMNP.

<table>
<thead>
<tr>
<th>Species</th>
<th>N (total)</th>
<th>Result</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Polygrammate herbraeicum</em></td>
<td>452</td>
<td>$X = 2.071$, df = 2, $P = 0.355$</td>
</tr>
<tr>
<td><em>Dolichomia olinalis</em></td>
<td>2264</td>
<td>$X = 231.315$, df = 2, $P &lt; 0.001$</td>
</tr>
<tr>
<td><em>Malacosoma americana</em></td>
<td>419</td>
<td>$X = 11.478$, df = 2, $P = 0.003$</td>
</tr>
<tr>
<td><em>Nadata gibbosa</em></td>
<td>177</td>
<td>$X = 0.029$, df = 2, $P = 0.985$</td>
</tr>
<tr>
<td><em>Halysidota tessellaris</em></td>
<td>1213</td>
<td>$X = 9.653$, df = 2, $P = 0.008$</td>
</tr>
<tr>
<td><em>Spilosoma congrua</em></td>
<td>513</td>
<td>$X = 3.788$, df = 2, $P = 0.151$</td>
</tr>
<tr>
<td><em>Idia americalis</em></td>
<td>233</td>
<td>$X = 10.566$, df = 2, $P = 0.001$ ^</td>
</tr>
<tr>
<td><em>Marimatha nigrofimbria</em></td>
<td>69</td>
<td>$X = 3.118$, df = 2, $P = 0.210$</td>
</tr>
</tbody>
</table>
Figure 1: Kaplan-Meier survival analysis curves for *Dolichomia olinalis*. The x-axis represents the number of trapping days, starting on the first trapping date of April 25th, 2014. The y-axis represents the proportion of cumulative moths at each location. There was a significant difference across all sights with the earliest emergence occurring at BSC.
Figure 2: Kaplan-Meier survival analysis curves for *Malacosoma americana*. The x-axis represents the number of trapping days, starting on the first trapping date of April 25th, 2014. The y-axis represents the proportion of cumulative moths at each location. There was a significant difference across all sights with the earliest emergence occurring at BSC.
Figure 3: Kaplan-Meier survival analysis curves for Halysidota tessellaris. The x-axis represents the number of trapping days, starting on the first trapping date of April 25th, 2014. The y-axis represents the proportion of cumulative moths at each location. There was a significant difference across all sights with the earliest emergence occurring at BSC.
Figure 4: Kaplan-Meier survival analysis curves for *Idia americanum*. The x-axis represents the number of trapping days, starting on the first trapping date of April 25th, 2014. The y-axis represents the proportion of cumulative moths at each location. There was a significant difference between BSC and RMNP with moths at BSC emerging earlier.