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**Editor's note:** This issue of the Purchase College Journal of Ecology was published during the COVID-19 pandemic when General Ecology was taught entirely remotely. I would like to commend the students for their hard work during this very difficult semester. Thank you for continuing to push yourselves to do innovative science and work together even when we could not be together. December 17, 2020

*Ollyson Jackson*

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# TREE SPECIES RICHNESS AND CIRCUMFERENCE VARY BETWEEN RURAL, SUBURBAN, AND URBAN ENVIRONMENTS

Amanda Salmoiraghi, Paola Cruz, Julie Gifford, Joshua Mickens

## ABSTRACT

*Urbanization has produced inevitable changes for ecosystems throughout northeastern regions of the United States. Many factors influence biodiversity in ecosystems, such as tree species richness. To study this, we collected data in rural, suburban, and urban forested landscapes across New York State, measuring tree species richness within a 15m by 15m plot of land. Our results show that rural sites had the highest average for tree species richness when compared to suburban, and rural sites. Urban sites, however, showed the highest average for tree circumference, while the averages for suburban data settle amid rural and urban. Our data potentially could lead to developing more effective forest management practices, mitigate the effects of climate change, and protect threatened native tree species.*

Keywords: biodiversity, deciduous trees, forested, urbanization, urban-rural gradient

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## INTRODUCTION

Urbanization, and by default, human interaction, have wide-reaching and lasting impacts on ecosystems (Cardinale et al. 2012). As people migrated into cities, terrestrial ecosystems were subsequently changed to fit the mold that humans needed for their living conditions. Humans influence a variety of environmental factors. Forest fragmentation as a factor of urbanization, the conversion of forest to other land uses, makes forests more susceptible to exotic species invasions, alters nutrient cycles, changes species composition, and affects tree species diversity (Evans and Perschel 2009). Disease can also be brought to new environments directly, or it can be influenced by pollution in the air, water, and soil. Insects and tree diseases can infest urban forests, potentially killing trees, and reducing the health, value, and sustainability of the urban forest (Nowak et al. 2018). However, studies show that tree plantings have been viewed as a key component to urban tree population's resilience to pests, diseases, and climate change (Cowett and Bassuk 2017). As a society, humans continue to grow, alter, and influence the world we live in, whether that be for better or worse.

Urbanization of land creates unavoidable environmental changes that ultimately influence biodiversity (Grimm et al. 2008). As movement into cities and creation of spaces for humans to populate increase, habitats, and ecosystems that were originally found there become fragmented and broken. This often creates edge sites, and it is difficult for species to live in these areas. This can potentially limit biodiversity and species richness in an area (Grimm et al. 2008). Urbanization can also directly affect

biodiversity through invasive species, soil imbalances, climate, water use in the area, and biogeochemical processes (Kowarik 2011). Studies on biodiversity in urban areas and parks have concluded that urbanization and human disturbance are some of the leading causes of a decline in biodiversity, but it has also been found that urban cities are found to have a high concentration of biodiversity (Nielsen et al. 2014). For example, urban forests have been found to have higher amounts of invasive and/or introduced species compared to rural forests (Kowarik 2011). An increase of introduced species could influence the overall biodiversity and species richness found in urban areas (Evans and Perschel 2009). Invasive species are well-positioned to take advantage of species diversity shifts because they tend to be site generalists, mature quickly, and have successful dispersal strategies (Evans and Perschel 2008). Our research can give more insight into the controversy over the effect of urbanization on biodiversity, species richness, and abundance. Forest composition and the environment have a relationship that has weakened; the implications of this have been noticeable to us and will become apparent even centuries later (Thompson et al. 2013).

As pollution continues to grow at an alarming rate, we see the product of climate change. Climate change is an ongoing issue that negatively impacts various ecosystems and can impact humans. If environmental conditions become harsher, species lacking the appropriate stress tolerance would be more likely to go extinct, while other species that have sufficient stress tolerance would likely survive (Marks et al. 2016). Increased severe weather patterns such as droughts, hurricanes, and forest fires can impact species richness and diversity in all types of environments, including forested areas in the rural to urban gradient (Evans and Perschel 2009). Since it ultimately affects the amount of biodiversity found in terrestrial landscapes, climate change is a major issue that conservation ecologists face. Climate change can influence the range of native trees and could potentially impact distribution and species richness (Kendall et al. 2011, Nitoslawski et al. 2016).

In the Northeastern United States, forest environments vary an incredible amount in biodiversity, species, and species richness across a rural to urban gradient. To better understand the differences in biodiversity between rural, suburban, and urban forested environments, we surveyed eight 15m by 15m plots and cataloged the species of trees found trunk circumference, and the number of trees within individual plots. We hypothesized that suburban sites would have greater biodiversity and species richness compared to urban or rural sites.

## METHODS

*Experiment setup.* Three different land-use sites (urban, suburban, and rural) were chosen to perform our tree abundance, diversity, and species analysis experiment. The collection sites were chosen on an urban to rural gradient and all within temperate and mostly deciduous environments typical in the Northeast, United States (Fig. 1). Data was collected Oct. 15-Nov. 4th, 2020. Leaf senescence occurs in the fall when the chlorophyll is lost in the leaves due to declining temperatures and decreased day length. This process was nearly complete at the rural collection site, the suburban collection site was at its peak, but only just beginning at the urban collection site. Urban collection sites included New York City as the most southerly at Pelham Bay Park in the Bronx, NY (PBPK). Suburban collection sites included a backyard landscape in Middletown, NY, and Five Islands Park in New Rochelle, NY. Rural collection sites were in Deposit, NY, located about 150 miles northwest of the urban site and westward of Catskill State Park. If needed, we requested permission from private landowners if the collection site was on their

property. Standard safety protocol was used: not sampling alone and wearing bright colors since it was hunting season. We chose collection sites with a common tree density for each of our land-use types. To begin the analysis, all four of us measured out two 15m x 15m plots (two plots for urban and rural, four plots for suburban) with standard 100' construction grade measuring tape with metric centimeter units on it. Each intersect was perpendicular to the last tape ensuring a square was being set up.



Figure 1. Aerial map of the NY region shows an overview of the data collection sites from the rural, suburban, and urban plots. Green stars (Urban Sites 1 & 2), yellow stars (Suburban Sites A & B, two sites each), and pink stars (Rural Sites 1 & 2). When only one star can be seen, it is because the plots were adjacent to each other.

*Data collection.* First, we began by walking the length of the plot along with the measuring tape. Each tree within the plot area was measured by circumference in centimeters at chest height. We continued to walk parallel to the transect tape from end to end, this ensured all trees within the plot were measured and recorded. Only the trees greater than 3 cm were recorded on a data sheet along with their species type. Resources used to identify tree species included iNaturalist and Google image search.

*Statistical analysis.* Aerial photos from Google Earth of each plot were provided for visual analysis (Fig. 2). As well as a regional area map with plots identified with a star (Fig. 1), Google Earth Pro was used for this. Upon completion of our data collection, we individually used Microsoft Excel to create box and whisker graphs to analyze the mean circumference of the trees measured per plot, including standard error. All individual collection site graphs were compiled for group data analysis by tree species richness by land use (Fig. 3A), tree species richness by plot (Fig. 3B), average tree circumference by collection site (Fig. 4A), average circumference by plot (Fig. 4B), biomass (total circumference added up in cm) of each plot (Fig. 5B), biomass of each per collection site (Fig. 5A). For the suburban collection sites, A & B were averaged.



Figure 2. Panel figures reflect Google Earth aerial photos of all rural, suburban, and urban plots.

RESULTS

Figure 3A demonstrates that the rural sites had the highest average tree species richness with an average of 4 different species per collection site. The suburban sites had an average of 3.75 species per collection site and the urban sites an average of 3.5 species per collection site. Figure 3B indicates suburban site 1-B had the overall lowest tree species richness with 2 per plot.

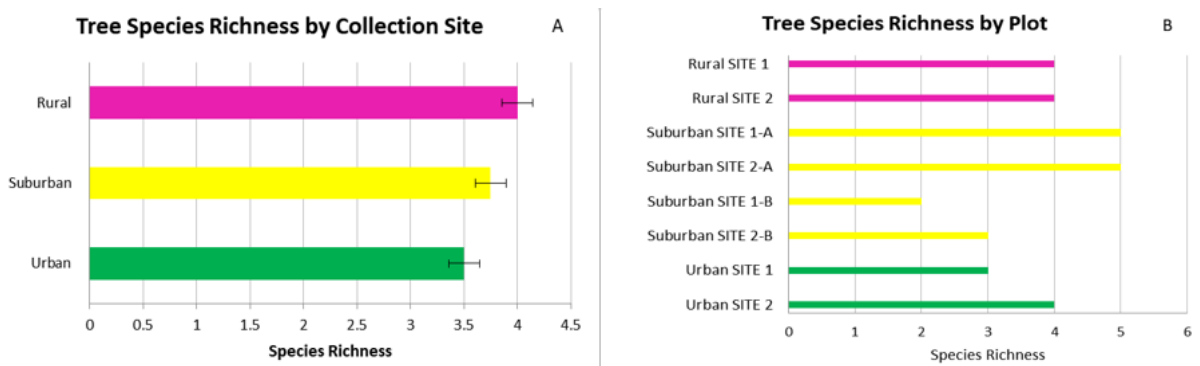


Figure 3. Tree Species Richness. A) Average tree species by collection site: rural had the most tree species richness while urban had the least. B) Tree species richness by plot: Suburban Sites 1A and 2A had the most species richness, while Suburban Site 1B had the least.

Our data reflected that the urban sites had the highest average tree circumference, 176 cm, compared to the rural and suburban. The suburban sites had an average of 96.26 cm and the rural site had an average of 65.07 cm. The rural sites had the overall smallest average tree circumference in cm. The difference between the plot means is demonstrated in Figure 4A, which shows that the difference between the urban sites and the other two sites is more significant than the difference between the suburban and rural sites. As depicted in Figure 4B, suburban site A had a lower average tree circumference, 55.92 cm, compared to suburban site B which had an average tree circumference of 136.61 cm, which is 80.69 cm more than suburban site A.

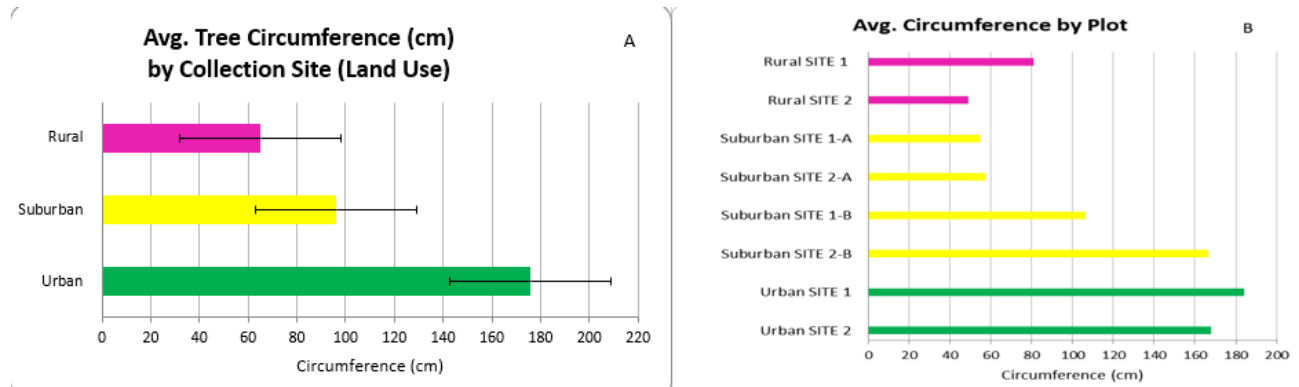


Figure 4. Average Tree Circumference in cm. A) Average tree circumference in cm by collection site. Urban has the highest circumference, while rural had the least. B) Average tree circumference in cm by plot. Urban Site 1 has the highest average circumference, while Rural Site 2 had the lowest.

Figure 5A shows the urban sample site had the highest total biomass, 1593.5 cm, out of all the sample sites. The rural site was close in the total biomass with 1536.53 cm. The suburban site had the lowest total biomass, 808.27 cm. Suburban site A had the overall lowest total biomass, 320.04 cm, which is seen in Figure 5B.

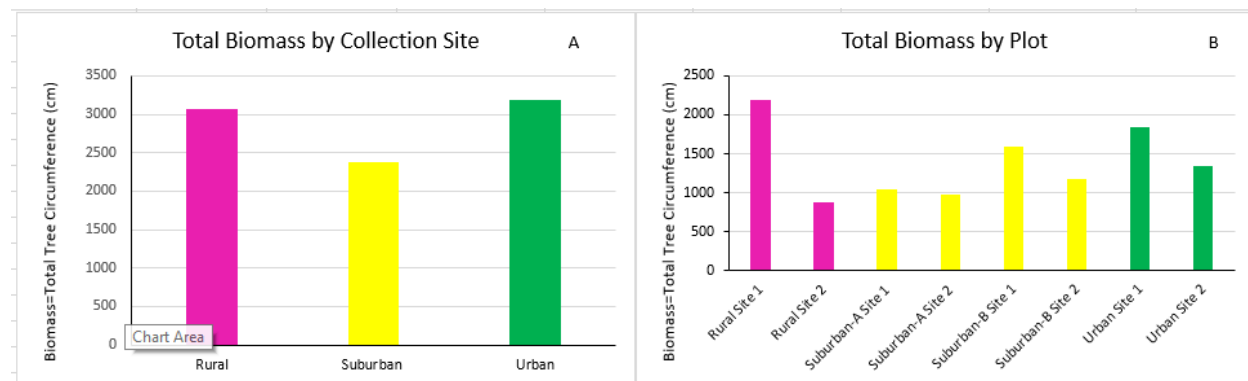


Figure 5. Total Biomass in cm. A) Total biomass in cm in rural, suburban, and urban sites. The Urban Site has the largest total biomass and suburban had the least. B) Total biomass in cm by plot. Rural Site 1 has the highest total biomass, and Rural Site 2 has the least.



## DISCUSSION

This study looked at the differences in tree species richness and abundance across the three types of communities: rural, suburban, and urban across New York State. We hypothesized that suburban collection sites would have a greater tree species richness and abundance than rural and urban sites. Because of a decline of biodiversity seen on a rural to urban gradient, we anticipated less tree species richness in urban areas. We predicted that the rural environment would not have the highest tree species richness because of logging in rural environments during recent decades. Our results showed that there is a higher amount of tree species richness and abundance in the rural site compared to the suburban and urban sites. Although the rural site had the highest tree species richness, there was only a minor difference between the tree species richness from all of the sites. There was a 0.25 difference in species richness between the rural and suburban and a 0.5 difference between the rural and urban sites. This data can possibly contribute to biodiversity and forest management studies and practices.

Many aspects could have affected the results of this experiment, which may have influenced the trends found in the data. Having two suburban sites could have impacted our results because we averaged the measurements between the two sites to be able to compare it to the rural and urban sites equally. When looking at the raw data, before averaging the two suburban sites, site A has on average the lowest scores in each measurement which could have affected the overall representation of the suburban category. Site A was located in a more northern part of New York, and site B was located closer to the urban areas of New York, almost near the urban site which is shown in Figure 1. According to Ricklefs and He (2016), there are three sources of variation that are statistically associated with tree species richness in a local sense: the biogeographic region, the sample size, and the local climate.

Our data collection sites are all located in the Northeastern United States, and the climate in the areas are relatively similar. All of the plots we surveyed were 15m by 15m, however, the number of trees sampled within those plots varied. Understandably, fewer trees would be sampled in urban environments when compared to a rural environment. The location for the Urban Site was collected in an area of PBPK in the Bronx, NY, where the trees could have been an anthropogenic addition to the park's landscape. That could be the cause of more habitat fragmentation because the trees could be strategically planted making the space between them much larger. That makes it much more difficult for biodiversity and species richness to thrive. There was a previous study conducted at the urban site, PBPK, by Robert DeCandido. DeCandido's study (2004) was a census of all the plant species found at PBPK conducted from 1994-1998, which was a follow-up to a previous study done by H.E. Alhes from 1947-1948. This follow-up study found that during 50 years the park lost 19.5% of its flora, and specifically lost 25.5% of native species and 12.8% of nonnative species, and also found that invasive species have settled at a rate of 2.7 species per year during that period. The results of his study highlight that as urbanization increases so does the decrease in species diversity. There is also an increase in invasive species which could account for some of the "biodiversity" observed in urban areas. Our study could be used to extend other investigations of urban areas to compare the flora from different periods.

While our preliminary research yielded interesting results, it would be suggested to conduct this research on a larger scale. The data collected in our research and the results concluded are useful to assess if biodiversity decreases as we move into the urban territory. In the future, we would increase the size of the plots surveyed from 15m by 15m, which could potentially further our understanding of the biodiversity of the area surveyed. There could be more species richness within a larger area. We would also increase the number of plots surveyed. A factor that influenced data collected was there were two

suburban data collection sites compared to one rural and one urban. This may have influenced the variation in our data. Our data was collected during the mid-autumn season, where many leaves were falling. This caused trouble in identifying trees in an area that had very few leaves left. A suggestion for future studies would be to conduct investigations during the late spring or summer months when the trees still have an abundance of leaves.

Having a larger sample size will help determine whether or not our hypothesis is being supported. Since our study was contained to the Northeast of the United States, future studies could expand upon our results by conducting surveys in other areas of the United States to see if the pattern continues. Studies performed in Japan by Yuta Kobayashi indicate the need for expansive research to be performed on the service of mixed-species forests. Their research concluded the ecological and economic benefits of mixed-forest management on a broad climatic scale. Indicating new evidence that mixed-species management is worthwhile to face climate uncertainty (Kobayashi and Mori 2017).

More variables could be added to the experiment to get a better insight into the biodiversity of the different locations because different factors can affect tree species diversity. Geography and climate, resource availability, and even income and economic status of homeowners in the area are some examples of drivers that can potentially affect tree species diversity (Nitoslawski et al. 2016). Additionally, species assemblages vary depending on overall ecosystem health and their proximity to a densely populated urban environment. The topography of an area can also greatly contribute to forest diversity (Diggins and Catterlin 2014). Incorporating sociological aspects could give a unique and more accurate view of an area's biodiversity. Light intensity, pH level, and levels of belowground organic matter are also factors that affect biodiversity. Measuring and comparing these factors for each location could lead to a better understanding of the ecosystem's services. It is important to study each component to qualify how biodiversity functions altogether, and how it is affected by each of these potential variables.

The results and expansion of this study can be useful for further investigation of biodiversity in different environments including rural, suburban, and urban. Biological systems have many components that influence each other and studying one component can help understand another. According to Cardinale et al. (2012), the efficiency of an ecological community is tied to its biodiversity; as biodiversity decreases, so does the efficiency of the community. Discovering and studying those factors can give insight on how to promote and protect biodiversity and community efficiency in these ecosystems. Research on urban biodiversity has become popular in the last few years due to the effects of urbanization on natural ecosystems. Recognition that urban ecosystems can be used in new ways to promote biodiversity has increased (Neilson et al. 2014). Our analysis could potentially help future studies investigating the urban forests controversy, as to whether or not urban forests have high biodiversity or not, and why. Focusing on promoting biodiversity in urban forests is a prospective form of land management to maintain the species richness.

Studies such as ours could contribute to conservation and biodiversity efforts. Doing this can also aid in the conservation of endangered species (Alvey 2006). Since climate change increases the likelihood of catastrophic rain events, recent findings indicate that tree diversity can reduce the growing risk of natural disasters such as landslides (Kobayashi and Mori 2017). Calculating the relationship between tree canopy cover and the density of stems has been found to predict tree species richness in urban forests (Gillespie et al. 2017). Investigating the predictors of species richness and knowing what affects the components that make up and influence the function of biological systems can aid in protection and restoration efforts. Our study can contribute to other studies that are looking into the individual components of biodiversity that affect one another and biodiversity as a whole.

## CONCLUSIONS

Our findings concluded tree species richness was the greatest at the rural collection site, there was <.5 species difference from suburban. Species richness was lowest in urban environments. However, the urban collection site had the greatest average tree circumference, with rural areas having the smallest. With urbanization increasing at significant rates, preserving forested land and old park systems is essential in maintaining green-space accessibility for urban populations (Evans and Perschel 2009). Protecting forested urban green space is a key factor in continuing to preserve urban ecosystem benefits. Through improved urban forest management practices and decreasing homogeneity in urban environments, such improvements can lead to increased resilience to non-native invasive species, improved ecosystem function, and improve the health and wellbeing of the entire community (Nock et al. 2016). The large biomass of an urban forested plot will have lasting impacts on mitigating the urban heat island effect and climate change. This data in conjunction with further carbon capture ability of deciduous forested areas on a rural to urban gradient will be crucial in proactively adapting to ever changing environments.

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## AUTHOR CONTRIBUTIONS

Conceptualization (all), Data Collection (all), Data Curation (all), Formal Analysis (all), Methodology (JG), Project Administration (AS), Resources (all), Visualization (JG, AS), Writing - Intro (AS), Writing - Methods (JG), Writing - Abstract (JM), Writing - Results (PC), Writing - Discussion (AS, PC), Writing - Conclusion (JG), Writing - review & editing (all).

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# **RUNOFF CONTAINING DISINFECTANTS BEING OVERUSED DURING THE COVID-19 PANDEMIC MAY HAVE NEGATIVE IMPACTS ON PLANT LIFE**

Sophie Bell, Gregory Gisser, Zachary Mitchell, and Joseph Zippelli

## **ABSTRACT**

*The current COVID-19 pandemic has led to many Americans using an excessive amount of disinfecting cleaning products. Panic amongst citizens and the need to protect human health resulted in an unprecedented amount of these chemicals being used. Chemical runoff of these products spread into the surrounding ecosystem threatening the local fauna and flora. To study the effects of disinfectants on plant life, we treated five different species of plants with hand sanitizer and bleach. Our results show that disinfecting substances have a highly negative effect on all plant life, greatly reducing the lifespan of the plants. Plants that were treated with the bleach solutions expired quicker than those treated with the hand sanitizer solutions. Therefore, we can hypothesize that chemical runoff due to the overuse of disinfectants during the COVID-19 pandemic has a detrimental effect on the environment.*

Keywords: Bleach, COVID-19, Disinfectants, Plants, Sanitizer

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## **INTRODUCTION**

Due to the current COVID-19 pandemic, many American citizens in rural and urban areas have been using an excess of cleaning products. Whether in residential homes, hospitals, stores, or factories, high amounts of chemicals being used will lead to the risk of chemical runoff affecting the environment. The extent in which chemicals are being used to combat the COVID-19 pandemic is unprecedented. The state of panic of American citizens and desire to protect human health results in overuse of certain cleaning products, which in turn poses a risk to human health and the environment. Exposure to these chemicals increases the risk of developing cancer, COPD, asthma, and other health conditions (Rai 2020).

It is well known that weather related run-off can cause chemicals to spread into the surrounding ecosystem, producing negative effects on the environment (Sakai et al. 2004). Studies from the early 2000's have proven that increased risk of chemical run off detrimentally affects more urban environments. This is believed to be true for a multitude of reasons, such as the vulnerable layout of urban environments as well as the increased use of chemicals in urban as opposed to rural areas. Furthermore, this study concluded that run off originating in urban environments is highly responsible for nutrient

pollution (Sakai et al. 2004). Additionally, it was concluded that common pharmaceuticals have been linked to endocrine disorders in certain organisms due to exposure through chemical runoff. These chemicals can penetrate the soil through sewage, bio-solids, and irrigation (Ohoro et al. 2019). The impacts of chemicals can be unexpected, and can occur even as a result of small concentrations (Russo et al. 2020). All species of plants are impacted by human actions, from urban woodlots to rare medicinal plants (Ganie et al. 2019, Fornal-Pieniak et al. 2019).

Hospitals and other medical facilities are also sources of toxic waste. Rao et al. (2004) believe that hospital waste poses a threat to public health and the environment. Rahman et al. (2020) agree and stress that due to COVID-19, excessive biomedical waste is the newest danger to public health and the environment. Scientists in Romania have explored the idea of waste flow worsening the pandemic. The possibility for virions combined with chemical waste from hospitals and medical facilities will undoubtedly cause damage to human life (Mihai 2020). Due to the severity of the pandemic, proper waste flow and monitoring has been deemed less important than other pressing issues surrounding COVID-19 in many countries across the globe. What gets ignored specifically, is the threat that improper waste flow poses to the environment (Mihai 2020).

In some parts of the world, large quantities of disinfectants are being applied onto roads, commercial, and residential areas to exterminate the COVID-19 virus. Such extensive use of disinfectants may kill beneficial species, which may then create an ecological imbalance (Rume and Didar-Ul Islam 2020). Russo et al. (2020) found that chemicals could disperse into the air, surface and groundwater which would then expose natural plants. They stress that ruderal, disturbance-tolerant plants that many insects rely on are also exposed to these chemicals.

The main substances that we are concerned with during this project are hand sanitizer and bleach, since these are the substances that are being overused the most during the COVID-19 pandemic. In fact, due to the public's need for these products, the FDA has decreased some of their regulations surrounding the content and production of hand sanitizers. High demand of disinfectants, combined with the WHO's advice of cleaning outdoor surfaces with these disinfectants, has ultimately led to their mass production and use in ways that could pose harm to the environment (Fairgrieve et al. 2020). Studying the impacts of such chemicals is important, as it is not uncommon for byproducts of certain production processes to actually be beneficial to plants, such as carbon byproduct materials (Kumar et al. 2020).

We aimed to explore in detail the effects of chemical run off and improper waste flow on natural plant life. It was hypothesized that our group would find evidence of a negative impact on natural plant life under simulated chemical runoff conditions.

## METHODS

To determine the effects of improper waste flow on natural plant life caused by COVID-19 disinfectants, our group conducted an experiment to see if several simulated waste flows would have an impact on the growth of common plants. To do so, each member of our group had to purchase five plants of the same species (plant species differed between members to cover a larger band of plant life) along with Clorox brand bleach and a 60%-80% alcohol content store bought hand sanitizer to be our disinfectants. The plants that we have acquired are as follows; Violets (*Viola*), Mums (*Chrysanthemum*), hostas (*Funkia*), and Kale (*Brassica oleracea* var. *sabellica*). Five plants were chosen so that there would be one plant designated to test each of the experimental groups: a 10% bleach solution, 50% bleach solution, 10% sanitizer solution, 50% sanitizer solution, and a control of plain water.

Each member of the group set up their individual plant species either indoors or outdoors (positions of plants varied between group members due to member locations). The mums were kept outdoors throughout the experiment, the kale and violets were kept indoors, and the hostas were outdoors,

until a frost warning was issued, at which point they were brought indoors. The plants, regardless of species/group members, were separately potted and labeled to keep track of each experimental group and to test the effects of disinfectant runoff simulated by watering the soil with Clorox bleach and hand sanitizer solutions. While runoff is not controlled in nature, each species of plant was watered on a “as needed” schedule. After the first full day of the disinfectant solution treatments, we began our daily data collection. We recorded the percent dead of each plant by comparing the amount of living vegetation to the dead/yellowed vegetation and was all to be recorded in an excel file for later analysis.

Each group member utilized the same, or very close to the same, materials. This included 5 individual plants of differing species per member. Every group member used Clorox brand bleach to ensure consistency; hand sanitizers ranging from 60%-80% isopropyl alcohol concentration were used in efforts to maintain consistency as well. Plain tap water was used as a control across each experiment. Further, a liquid volume measuring tool was used to create each solution and ensure that ratios were consistent across experiments. Those recording height utilized centimeter rulers to mark their findings. All photos presented were taken on each members’ individual cameras. Lastly, an excel file was used to record and share all collected data over the course of the experimental period.

## RESULTS

After performing a paired t-test, we found a significant difference before and after applying the 10% hand sanitizer, 50% hand sanitizer, 10% bleach, and 50% bleach treatments ( $p < 0.05$  for all treatments). We did not find a significant difference before and after treating the plants with water ( $p = 0.14$ ).

As expected, the 50% bleach and hand sanitizer solutions had larger impacts on the plants than their 10% counterparts. Most of the control plants didn’t wilt much, with the exception of the hostas. This was likely due to overwatering or other factors. None of the plants seemed to be more resistant to the effects of the cleaning chemicals than any of the others.

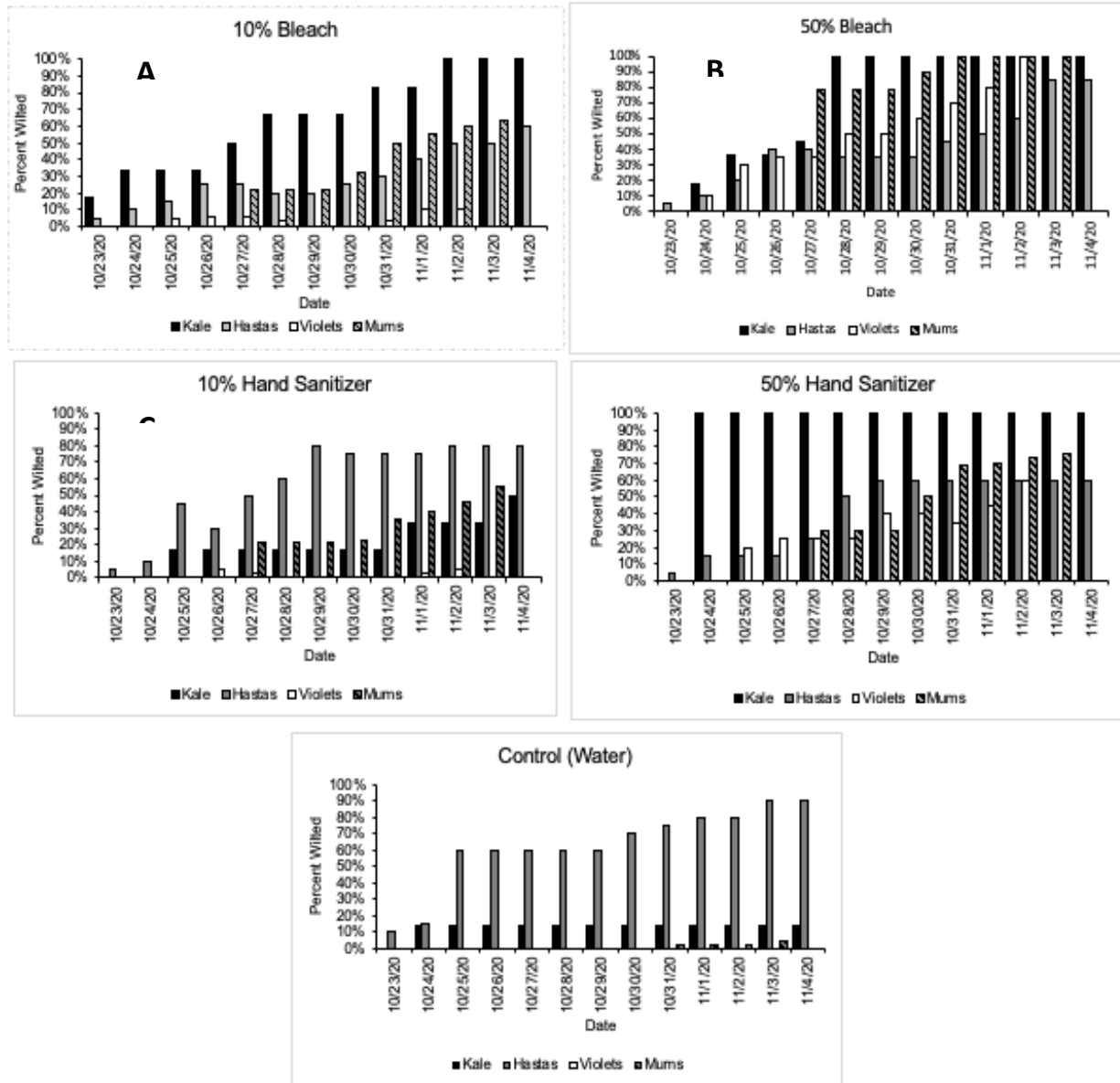


Figure 1. A) The death rate of kale, hostas, violets, and mums over a period of two weeks. Each plant was watered with a ratio of 10% bleach to 90% water. B) The death rate of each plant using a ratio of 50% bleach to 50% water. C) The death rate of each plant using a ratio of 10% sanitizer to 90% water. D) The death rate of each plant using a ratio of 50% sanitizer to 50% water. E) The death rate of each control plant over a period of two weeks.



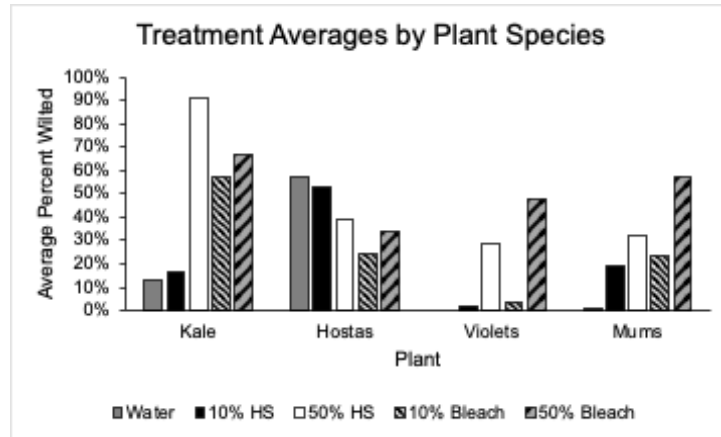


Figure 2. Average percent of plant wilted for each plant species and treatment.

## DISCUSSION

We hypothesized that the simulated chemical runoff would have a negative effect on natural plant life. We focused on the two most common disinfectants being overused during the COVID-19 pandemic, hand sanitizer and bleach. We tested our hypothesis on four different sets of plants that we each treated individually. Although hand sanitizer did affect the plants, it did so at a slower rate than bleach. As expected, it was noted that the percentage of chemicals in our mixtures had a great impact on the rate of effect as well. Plants treated with a lower dosage of bleach and hand sanitizer showed detrimental effects throughout experimentation in most cases, but not to the same degree as those treated with a higher dosage of the chemicals. This is similar to a study performed by Jackman and Hughes (2010) on trihalomethanes (THMs) that were found in the soil and groundwater at sites where the release of organic solvents had occurred. Jackman and Hughes (2010) explain that THMs often contain sodium hypochlorite, otherwise commonly known as bleach. THMs may also contain chloroform, which is produced by mixing bleach and alcohol. Alcohol is the main ingredient in hand sanitizer and mixing it with bleach would cause a dangerous reaction. Chloroform is toxic when inhaled or comes into contact with the skin (Rai 2020). Therefore, THM's in soil would have a detrimental effect on the plants they come into contact with.

The fact that the chemicals affected the life span of the plants was expected as Ohoro et al. (2019) stressed that contamination in soil may lead to a build-up in plants. Additionally, they believe that this



Figure 3. The effects of bleach and hand sanitizer on garden mums.

concentration of chemicals may expose animals and humans through the ingestion of plant material. There has been an understanding amongst the public and scientists that the everyday products we use can have inadvertent and lasting effects other than their intended results. A 2012 study found that monitoring the contents and presence of pharmaceuticals in surface water is a viable way to track anthropogenic influence on the environment (Vystavna et al. 2012). This in essence proved our manipulations to be a good way of testing potential runoff effects on natural plant life.

One concern that arose while conducting our post analysis was the method in which each experimenter treated their plants. For instance, while we all used treatments composed of the same chemicals, and the same ratios of those chemicals, it should be noted that they were not exactly the same. Due to the manner in which we had to conduct our experiment, we had to perform our manipulations in different geographical areas. This made it difficult to obtain the same materials. Each experimenter used bleach of the same brand and concentration. However, the percentage of isopropyl alcohol varied slightly among the types of hand sanitizer used in each experiment. We all used a hand sanitizer with an isopropyl alcohol range from 60% to 80%. If we were to conduct the same experiment again, this is one parameter that we would change, in order to ensure our treatments were more consistent across each plant species.

Another concern we had was the location of our plants. Two experimenters kept and monitored their plants indoors, near sunlight, as directed for each respective species of plant. One experimenter kept and monitored their plants outside for a short period of the experiment, and then moved the plants indoors. One experimenter kept and monitored their plants outdoors for the duration of the experiment. Additionally, some plants are hardier than others such as the violets and garden mums. The Hosta control plant was close to death by the end of the experiment, possibly due to overwatering. These factors would be edited if we were to reconduct this experiment, to make sure that the results we obtained were not impacted by any outside variables we were not studying.

One interesting observation that was made during the course of this experiment was the impact bleach had on the soil. In the kale plants that were treated with bleach, the soil turned a yellow color, and the liquid that drained out of the bottom of the plants was black. It is likely that this was simply coloration of the soil being eroded by the bleach, however it could be indicative of a more serious impact to soil health and should be studied further if possible. Some factories that produce bleach are guilty of producing toxic chemical runoff such as a bleach kraft pulp mill. The bleaching process uses chlorine bleach which produces chlorophenol compounds that are discharged into the environment. These compounds are toxic environmental pollutants (Dominguez et al. 2002). We can theorize that other factories that produce chlorine bleach may also leak this toxic chemical into the soil and groundwater located near the site, harming the local flora and fauna. It is interesting to note that although chemical runoff due to COVID-19 negatively affects the environment, air pollution has decreased. Berman and Ebisu (2020) found that due to decreased vehicle and airline travel, our air quality has substantially improved. This finding is extremely important as exposure to poor air quality increases the risk of a COVID-19

Eventually, the COVID-19 pandemic will come to an end but the issue of contaminated soil in urban environments will continue. Approximately 55% of the world's population live in urban areas which is expected to increase. Urban soil is the most chemically contaminated soil and lacks fertility (Kumar et al. 2020). Fornal-Pieniak et al. (2019) stress that anthropogenic factors such as pollution, increasing temperature, and soil compaction all affect plant diversity. They believe it is imperative for urban areas to protect natural habitats as they maintain biodiversity (2019). Ganie et al. (2019) agree and stress that management interventions to conserve threatened species should be implemented after first assessing for operative threats. Regulations to control chemical runoff and other factors that affect the surrounding environment need to be created and enforced in order to protect natural plant life and the wildlife that depend on them.

## CONCLUSIONS

Ultimately, the impacts of chemical runoff as a result of increased chemical use during the COVID-19 pandemic requires more research, especially since the pandemic is not over. Additionally, the likelihood of these chemicals entering the environment remains unknown, but possible. Further studies could look into the impacts of developing a new vaccine on the environment, as that is an incredibly complicated process requiring many different chemicals and materials that may accidentally end up in the environment.

Overall, our hypothesis was supported by the data, and the simulated chemical runoff had overall negative impacts on the plants. We were unable to study long term impacts as a result of our short time frame, which is why the concentrations were so high. Continuing to study the effects of common chemicals being used during the pandemic is incredibly important so that there are not unintended consequences to our environment. Additionally, the impacts of the chemicals on soil health may be important, as many odd and intriguing changes in the soil did occur.

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## AUTHOR CONTRIBUTIONS

Conceptualization (all), Data Collection (all), Data Curation (all), Formal Analysis (ZM), Methodology (all), Project Administration (all), Resources (all), Visualization (all), Writing: Abstract (GG), Introduction (SB, GG), Methods (JZ), Results (ZM), Discussion (SB,GG), Conclusion (ZM), Acknowledgements (GG), Writing: review and editing (all)

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## URBAN BIRDS HAVE SHORTER FLIGHT INITIATION DISTANCES (FIDs) THAN RURAL COUNTERPARTS

Yesllen Godoy, Maria Molina, Erika Seidman, and Kyra Cooleen

### ABSTRACT

*Human disturbances, as a result of urbanization, interfere with animals' natural habitats. Animals such as birds must habituate in order to survive; they do this by displaying certain behavioral syndromes that allow them to adapt and succeed. Flight initiation distances (FID) can be used to quantitatively measure how tolerant birds are to humans before they feel threatened and fly away. To test whether birds from urban environments display greater levels of boldness, we measured flight initiation distances in both urban and rural environments. We found that birds from urban environments had shorter FIDs, indicating more boldness compared to rural birds. In both urban and rural habitats, the more days we observed the birds the shorter the FIDs were, suggesting that birds become more habituated to humans over time. This study did not take different bird species into account, rather the difference between rural and urban bird FIDs as the main focus. Our study concludes that birds are bolder as a result of human disturbances in urban environments.*

Keywords: Birds; Boldness; Flight initiation distance; Rural; Urban

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### INTRODUCTION

Human population greatly affects the overall wellbeing of wildlife ecosystems. The increasing amount of urbanization seen throughout the world drives all species to adapt to their ever-changing environment (Clucas 2012). Human disturbances have not only changed the environment, but also the species that live within it. Although there are varying population sizes used to distinguish between rural and urban densities, the population size used for this experiment to signify an urban area is over 50,000 people and a rural area is a population consisting of 2,500 people.

Humans have forced species to adapt and develop various methods to survive in densely populated and industrialized areas. Urban species have different behavioral syndromes than those of their rural counterparts; some may include changes in foraging, mating, or territorial responses (Tsurim 2008). If a species resides in an urban environment, it may have reduced sensitivity to vehicular traffic or be more habituated to people. Species residing in more rural locations may show increased protection over their territory or a heightened awareness of predators. A behavioral syndrome that is most likely to vary between urban and rural species is boldness. Boldness is defined as the tendency of an animal to engage in risky behavior (Atwell et al. 2012) and is seen in various aspects of the bird's behavior (i.e., how close it lands to something/distance it makes it fly away). Boldness can also be related to genetic plasticity, the ability of multiple phenotypes to be produced by a single genotype and is shown to have phenotypic

variability between species. Depending on the species in question, a bird's boldness intensity can vary. This variation can be seen within the same species or across different species, as well as in varying bird sizes (Bonier et al. 2007).

This experiment aimed to understand the behavioral changes seen in birds due to the impacts of urbanization. An indicator that can quantitatively measure a bird's boldness is flight initiation distance (FID), which is the distance between the source of disturbance and the bird at the point where it flies away in response. FID can be related to a bird's anti-predation awareness along with their confidence to detect any potential predators (Zhou and Liang 2020). Previous studies have indicated that larger bird species have increased flight initiation distances due to the increased amount of energy needed to fly away (Møiler 2008 and Blumstein et al. 2005). Other studies using FID as a unit of measurement have indicated a possible reduction in sensitivity to stressors in urban conditions (Mikula 2014). A bird's alertness to a potential threat can be associated with the startling distance; this is a distance at which a bird becomes aware of a predator. Therefore, a higher startling distance is positively related to FID (Cooper 2005). However, just as FID is unique to each species, so is the startling distance (Blumstein 2003).

For this experiment, we observed birds and measured their boldness in four locations with different population densities, two being rural and two being urban. It is hypothesized that birds in denser urban areas will have lower FIDs, indicating that they are bolder than their rural counterparts. Based on the location in which the birds were observed, we can hypothesize that birds who reside in more urban areas have a shorter FID than those of rural locations, which is indicative of greater boldness.

## METHODS

**Study Area.** We conducted the study from October 21st, 2020 to November 3rd, 2020. To compare boldness in birds between urban and rural sites, we used two urban populations and two rural populations. The two urban environments used in the study were New Rochelle, NY consisting of 78,742 people and Franklin Square, NY consisting of 32,329 people. Putnam Valley, NY with 11,600 people and Hopewell Junction, NY with 454 people were considered rural areas (U.S. Census Bureau QuickFacts: United States). For this study, we considered Franklin Square to be an urban area.

**Flight-initiation distance.** FID measurements were taken throughout the day, for about an hour to an hour and a half. The observation site consisted of a bird feeder filled with bird food in either a backyard or a front yard of the observer's home. For each site, we used the same brand of bird food, it consisted of black oil sunflowers (*Helianthus annuus*), cracked corn (*Zea mays*), millet (*Pennisetum glaucum*) and milo (*Sorghum bicolor*). To measure FID, a metric tape was placed from the bird feeder and extended to where the observer was standing. The observer then started at the 0-foot mark and walked alongside the metric tape at a slow pace. At the moment the bird flew away, we recorded the distance between the observer and the startled bird. This

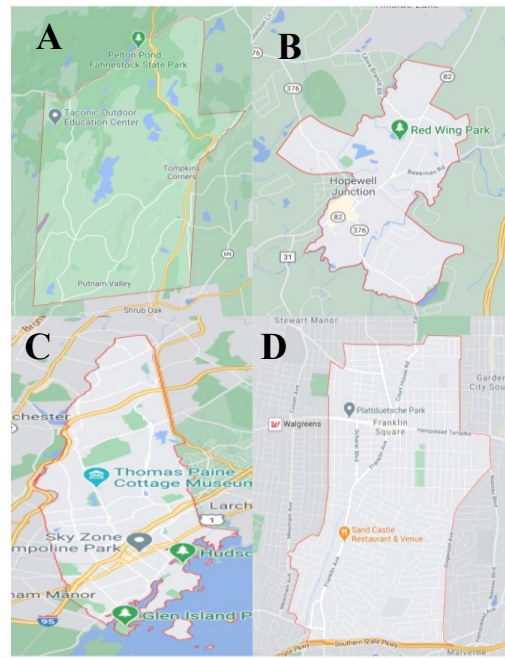


Figure 1. Map A represents Putnam Valley and map B represents Hopewell Junction. Map C represents New Rochelle and map D represents Franklin Square. Maps A and B display our rural sites and maps C and D represent our urban sites.

was repeated for each bird that was observed at each site for the three weeks. Over the three weeks, there were ten trials conducted in the suburban setting and eight conducted in the urban setting.

## RESULTS

The four locations used in this experiment were plotted, via Microsoft Excel, based on their population to further understand their size differences (Fig. 2). The most urban location, New Rochelle, is approximately 2.5 times larger than Franklin Square and is approximately 175 times larger than the smallest rural town, Hopewell Junction. There was an inverse correlation between the average FID and population size found; the average FID has been shown to increase as the population size decreased (Fig. 3). The smallest rural town, Hopewell Junction, had an FID average of 18.04 feet, while the largest urban town, New Rochelle, had an FID average of 15.33 feet. The average FID for both rural towns was overall lower than the average for the urban towns (Fig. 4). This demonstrates that urbanized areas have bolder birds.

The daily FID for each location shows variability in the data, however, all but one location represents a decrease in FID (Fig. 4); this is represented by the negative slope. All locations, except Franklin Square, have a decreased FID, although for this location there is a substantial outlier that is likely to alter the values. Putnam Valley had the greatest change in FID, shown by the lowest FID of 10 feet and the highest FID of 36 feet. Hopewell Junction had the lowest change in FID, due to the shortest FID being 12 feet and the longest FID being 22 feet. The decreases in FID are most likely due to the amount of urbanization in the bird's respective location and/or how habituated to the observer the bird became. The total average FID per location shows a relationship between location and the average distance at which the birds flew away (Fig. 5). The outliers at each site can explain environmental differences such as a new bird coming to the feeder or if the color of our clothes was potentially alarming to the birds deterring their presence.

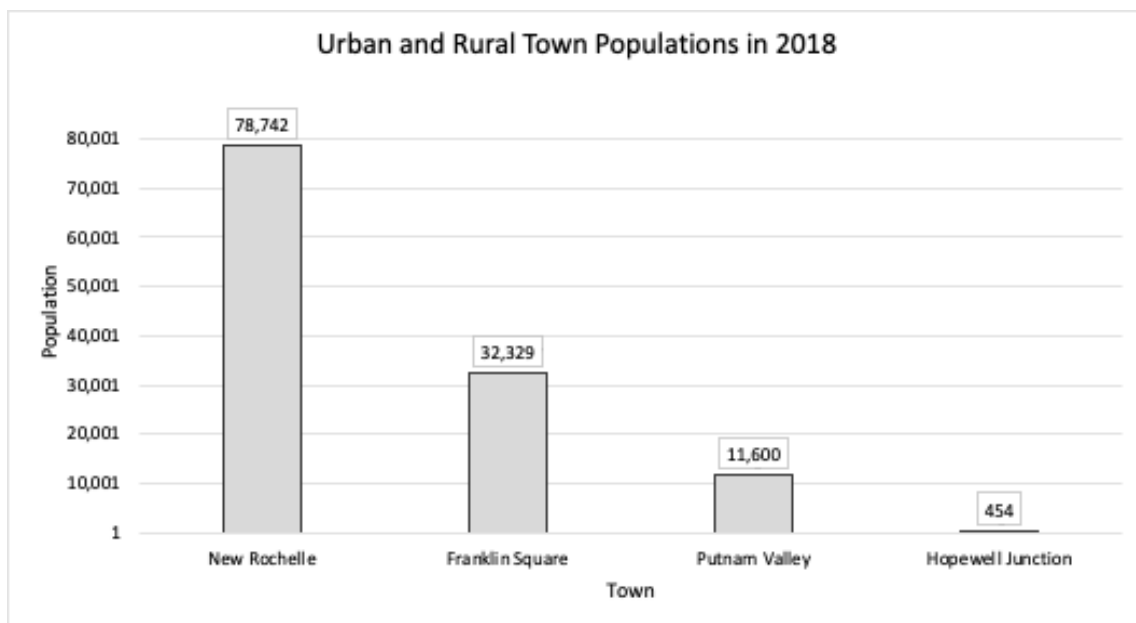


Figure 2. Population sizes for each town used in this experiment.

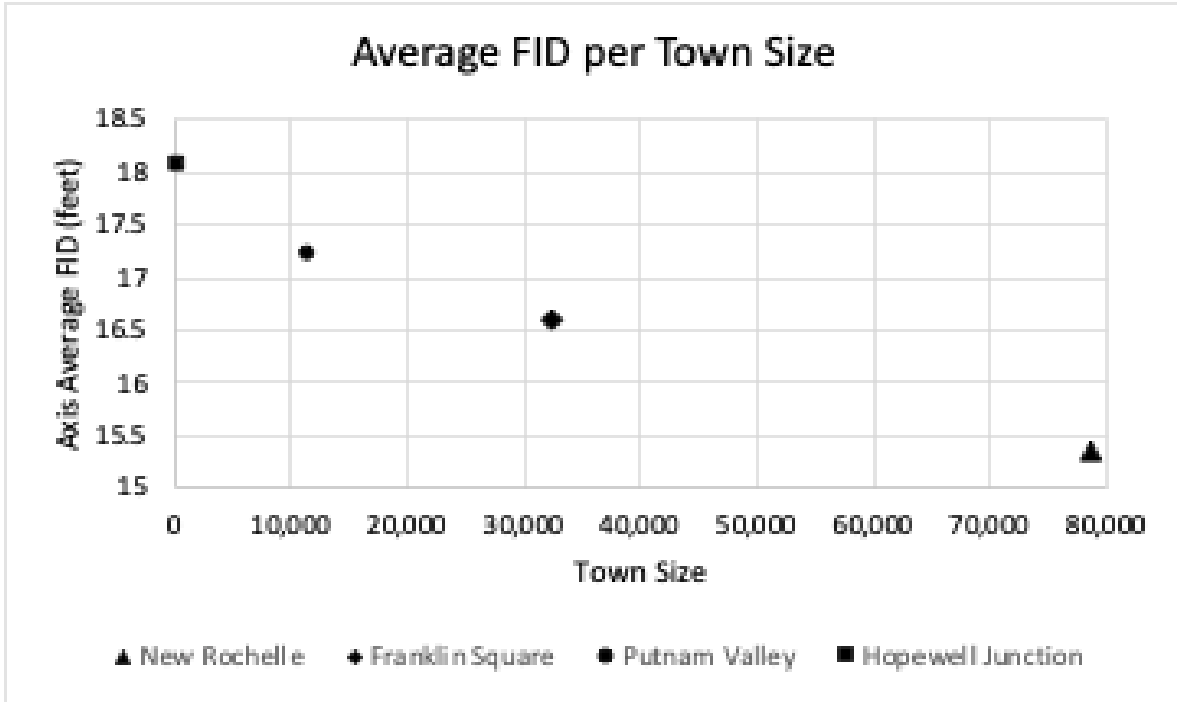


Figure 4. Average FID for the town size in which they were recorded. This shows the decrease in FID as the population size decreases.

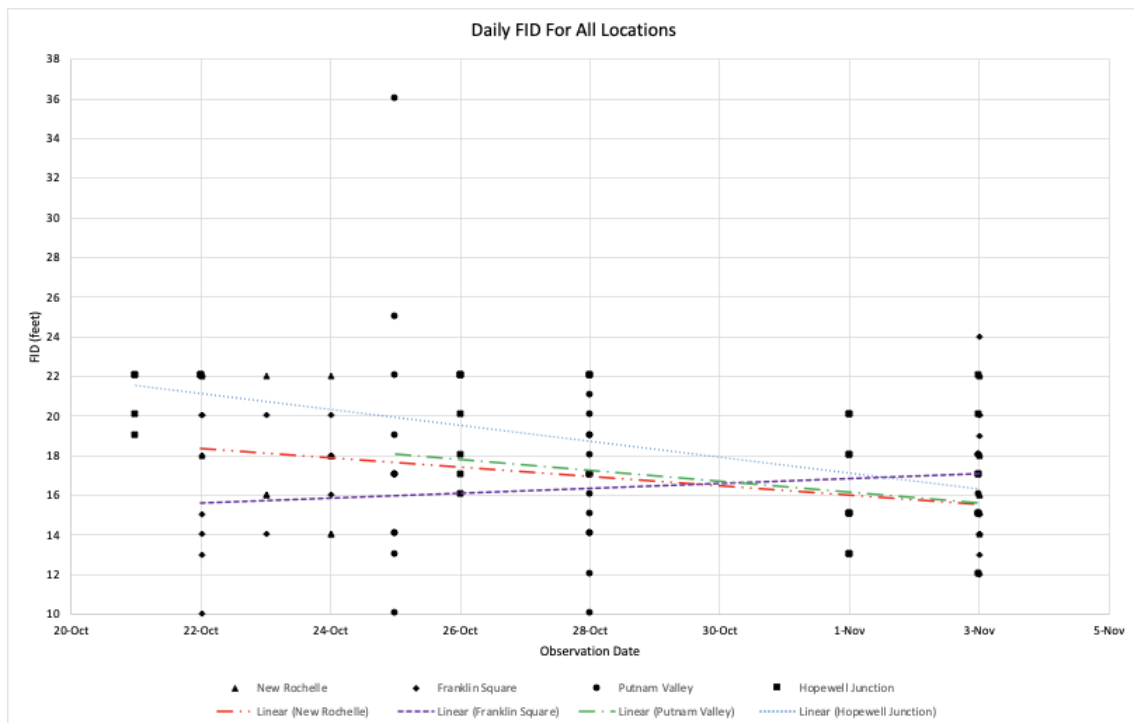


Figure 3. The daily FID for each location with a linear trendline for each location. All towns (except Franklin Square) show a decrease in FID, which is most likely due to the birds becoming desensitized to the observer's presence. The outlier on October 22nd, for Franklin Square, shows that the observer effect is not always true.



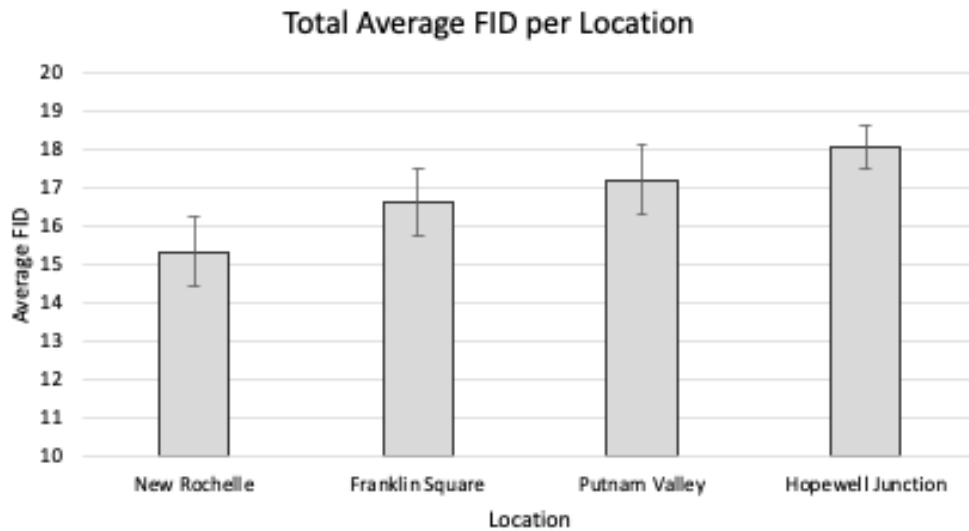


Figure 5. The average FIDs for each location with standard error bars representing the gradual increase in FID as the population size decreases.

## DISCUSSION

In this study, we examined the boldness of birds in urban and rural areas, to test whether living in a more urbanized population causes birds to be bolder. Using flight initiation distance (FID) as a method of measurement, we compared data between four towns, two urban and two rural. New Rochelle and Franklin Square represented our urban towns. Putnam Valley and Hopewell Junction represented our rural towns. From the data collected, we were able to observe that urban birds had a shorter flight initiation distance compared to rural birds. This led us to believe that urban birds are also bolder than rural birds which can be caused by different factors. Boldness and disturbance tolerance are defined as the distance at which the animal moves to avoid the threat, they feel from the approaching human observer (Hall et al. 2020). Urban bird species having a shorter FID can be related to their desensitization toward external stimuli. Rural birds are more likely to be afraid of humans and will fly away at further distances which is shown by rural birds having a lower FID. Our results indicated that urban birds are bolder due to higher levels of disturbance and human exposure.

The main trend we observed from our results was the shorter flight initiation distances in birds that live in more urbanized areas. We observed that the average FID for our urban towns was shorter than in rural towns. A reason for this trend may be due to natural selection or it was achieved through phenotypic plasticity, which would allow for rapid behavioral adjustment to human disturbance (Lill and Vines 2015). This trend also led us to believe that urban birds are bolder than rural birds. Other studies have shown that urban birds know humans do not usually pose a threat resulting in the birds having a shorter FID (Vincze et al. 2016). It can also be due to factors such as sex, size, and age of the birds. We would expect shorter flight initiation distances because FID is supposed to decrease with an increasing amount of time that the species occupies urban environments (Davey 2019). Another trend that was noted is that as the observer continued to go to their observation site, the flight initiation distances decreased. This can be due to the birds becoming more accustomed to the observer (Dingemanse et al. 2010), which has led to a decrease in responsiveness to the FID test.

While conducting our study, we did not consider the kind of birds in our data collection sites and how that could have affected the birds' flight initiation distances. Boldness can be a species-specific trait that could have had an impact on our results (Evans et al. 2010). The size of the bird could also have had

an influence on the FID. According to Blumstein et al. (2005), bigger birds had larger flight initiation. Because we did not consider the different bird species in our data collection sites, we do not know how that could have influenced boldness. We also did not consider how the color of our clothes could have had an effect on flight initiation distances. According to the species confidence hypothesis, birds are more attracted to colors that resemble their feather, so it is possible that FID could be impacted by the color of the data collectors' clothing (Zou and Liang, 2020). Future experiments in this field should track the species of birds in their area in order to determine if there is a trend of FID values with different birds. It can also be beneficial for future work to consider the kind of clothes worn by the data collectors to make sure there is no influence on birds' flight initiation distance.

Urbanization as well as human disturbances are factors that affect the natural habitats of animals. If animals are unable to adapt to selective pressures in their environments this can cause a species to become extinct due to natural selection. Atwell et al. (2012) found that human disturbances in urban areas have led to the fast evolution of increased boldness. Therefore, it is important for animals, like birds in this case, to mitigate human disturbances in more urbanized environments in order to succeed and reproduce.

## CONCLUSION

Determining if human population density affects bird behavior can be beneficial to our understanding of the human impact on the natural environment. For our study, we observed how close birds in areas of different population densities allowed an observer to approach before flying away. Our findings indicated a significant difference in flight initiation distances (FIDs) between birds observed in rural and urban areas. The differences in behavior by location confirm our hypothesis, stating that birds in more urban areas would have shorter FIDs than their rural counterparts. These urbanized birds demonstrate more boldness and less sensitivity to external stimuli. This conclusion helps us to understand the impact of human disturbances on birds and their behaviors. By comparing the behaviors across different populations, we can determine the effect of different stimuli on a population. Future studies wanting to further develop this research should consider species when collecting data, in order to obtain more accurate information. Another consideration may be to test different stimuli, other than a human approaching the food source. Along with understanding the impact of humans on bird populations, this data can be used to determine how successful a population of birds may be if introduced and how we can conserve populations in developing areas.

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## AUTHOR CONTRIBUTIONS

Conceptualization (MM, YG, ES, KC), Data Collection (MM, YG, ES, KC), Data Curation (MM, YG, ES, KC), Formal Analysis (ES), Methodology (MM, YG, ES, KC), Project Administration (MM, YG, ES, KC), Resources (MM, YG, ES, KC), Visualization (MM, YG, ES, KC), Writing (introduction) (ES), Writing (discussion) (MM, YG), Writing (abstract) (YG), Writing (methods) (MM), Writing (results) (ES), Writing (conclusion) (KC), Writing (review and editing) (MM, YG, ES, KC).

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# POLLINATORS PREFER NATIVE PLANTS OVER NON-NATIVE PLANTS

Maryrose Fabrizio, AJ Jacobus-Oseroff, Gabrielle Mendez, Kyle Starkey

## ABSTRACT

*Non-native plants have detrimental effects on native ecosystems. This can subsequently have a cascading effect on the feeding habits of pollinators. The effects non-native plants have on native plants are clearer; but the relationship between non-native plants and pollinators is more ambiguous. To study this, we went to various parks throughout New York and observed pollinator's habits in reference to preferring native or non-native plants. We also took note of the ambient temperature to see if it affected pollinator activity. The results in our trials concluded that pollinators preferred native plants over non-native plants. We also observed more pollinator sightings during warmer weather trials. Therefore, we can hypothesize that pollinators prefer native plants over invasive plants.*

Keywords: ecosystem; invasive alien plant (IAP); native plant; non-native plant; pollinator; preference; plant-pollinator interactions

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## INTRODUCTION

Human interaction has changed the landscape of Earth by using it to accommodate our own needs. Expansive development and urbanization have caused fragmentation in ecosystems as humans have expanded on undeveloped land and disrupted the natural way of life. By shaping the natural environment to match human needs, we have changed land use in ways that have benefited the human population, at the expense of native plants and animals. Destruction and fragmentation of habitat continues to cause mass displacement of plants and animals (Alarcón and Burkle 2011, Goulson et al. 2015, Harrison and Winfree 2015). This habitat fragmentation puts stress on the ecosystem and those in the surrounding area. As humans populate areas, we bring harmful factors like non-native species, pollutants, and disease that further disrupt native populations (Cane and Tepedino 2001, Goulson et al. 2015, and Lowenstein et al. 2019). These environmental disruptions cause changes in ecosystem structures and dynamics, directly affecting pollinator networks (Valdovinos et al. 2009).

With the introduction of non-native plant species, it's important to understand how these non-native species affect the native environment. Once established, non-native plant species become important to the network structure of plant-pollinator interactions (Valdovinos et al. 2009). Non-native plants are often able to germinate earlier and spread more efficiently than native plant species (Chrobock et al. 2011), increasing their abundance, therefore changing ecosystem networks. Habitat resource availability dictates pollinator action.

Pollinators play a significant role in spreading seeds and pollen of plants. The increasing abundance of non-native plants has decreased the proportion of native to non-native species across global landscapes. To make effective native pollinator conservation measures, it is important to know how they respond to the plants that are available to them across changing landscapes. There is evidence (Harrison and Winfree 2015, Lowenstein et al. 2019, Staab 2020) that supports the optimal foraging theory which predicts changes in foraging behavior of animals (Pyke 1980). This theory helps to suggest that pollinators on man-made landscapes, such as parks, are going to behave differently than pollinators on untouched land. A pollinator's response depends on the resources available to them. If a habitat becomes fragmented, a pollinator spends more time in one place but will travel to less places because of scarce availability of resources (Harrison and Winfree 2015).

As we develop man-made landscapes, we shrink natural habitats and introduce non-native species. Community development, through garden and landscape management choice influence the make-up of the surrounding areas of remaining natural habitat. Though the exact impacts of these decisions are unclear, we do know that bringing non-native species aids in their spread onto natural habitats (Knapp et al. 2012, Lowenstein and Minor 2016, Lowenstein et al. 2019). This causes a domino effect, increasing competition for resources such as nutrients, light, space, and water (Dietzsch et al. 2011, McKinney and Goodell 2011), and ultimately loss of plant species diversity (Bartomeus et al. 2008). Loss of diversity directly affects plant-pollinator relationships. The introduction of a new species gives the pollinator more than one option, and the non-native species will either overtake or coexist with the native species. Non-native plants can hinder native plant reproduction through their pollen introduction, causing pollinators to carry mixed pollen loads and deposit pollen of the non-native species instead of the pollen needed by the native plants (Larson 2008).

Non-native plants tend to be heartier and germinate earlier, enabling them to grow and thrive in new environments (Chrobock et al. 2011). The porcelain berry (*Ampelopsis brevipedunculata*), a deciduous perennial vine native to northeast Asia, is a great example of how non-native species introduction can disrupt native habitats. It was brought to the United States in the 1870s as a landscape plant. The berry has persisted to invade in twelve states, due to its heartiness and pest resistance (PCA 2005). Spread of non-native landscape plants like the porcelain berry affect pollinator behavior by altering their available resources (Gillespie and Elle 2018). Changes in weather patterns can exacerbate the changes further because even in inclement weather or climate change, non-native plants can still survive and adapt, giving them more space to spread.

The goal of our study is to distinguish if pollinators prefer native species over invasive species in city parks. Study of pollinator actions is vital to the conservation of species in plant-pollinator networks. The plant preferences of pollinators have been studied to varying degrees with mixed results. Pollinators will freely visit non-native plants (King and Sargent 2012, Lowenstein et al. 2019), especially in urban environments, where native plants may be scarce (King and Sargent 2012, Harrison and Winfree 2015, Staab 2020). When both native and non-native plants are available, pollinators prefer native and near-native cultivars, but pollinator preference has more to do with plant characteristics than origins, though most have been focused on urban and suburban neighborhoods and rural areas (Lowenstein and Minor 2016, Tiedeken et al. 2016, Lowenstein et al. 2019). Pollinator visitation has been shown to decrease across the urban gradient in numerous studies (Williams et al. 2011, Chrobock et al. 2013, Harrison and Winfree 2015). Several studies have found that pollinator preference has more to do with plant characteristics than origins, though most have been focused on urban and suburban neighborhoods and rural areas. By studying pollinator activity in city parks, we hope to contribute to the understanding of how community development of semi-landscaped plots impacts pollinator preference.

## METHODS

*Field site.* For our sites we chose places that would have a diverse amount of invasive and native plants. We decided that parks are a great representation of the different vectors that spread invasive

plants; there are also many examples of native plants at parks as well. We then proceeded to record our data in various parks located in New York State. In total we made 40 observations: 12 at Hutton Park in Kingston (Ulster County), 10 at Blue Mountain Reservation in Peekskill (Westchester County), 6 at Downing Park in Yorktown Heights (Westchester County), and 12 at Kissena Park in Queens, New York City (Fig. 1). All of our data was collected between October 25 and November 7, 2020, a time when pollinator activity usually decreases due to the impending winter. Recording temperature was important to see the relationship between pollinator prevalence and temperature. The instrumental observation of this study was to examine if pollinators were on a native plant or a non-native plant. We would try to find an equal amount of invasive and native plants in each trial, and we would observe the plants for 20 minutes to record pollinator activity. We would take note of the plant species type, the pollinator species type, and the temperature. Upon arriving home, we would insert our data onto a spreadsheet in Google Sheets and Microsoft Excel. The tool that we used to identify the plant species and pollinators was the application iNaturalist on iOS devices. The application was instrumental in our ability to accurately identify the exact plant and pollinator species in our study.



Figure 1. The locations where we made our observations are depicted in the map above. Hutton Park in Kingston was the northernmost point; Kissena Park in Queens was the southernmost point (113 miles apart). The two locations in the middle were Blue Mountain Reservation in Peekskill, and Downing Park in Yorktown Heights (6 miles apart).

## RESULTS

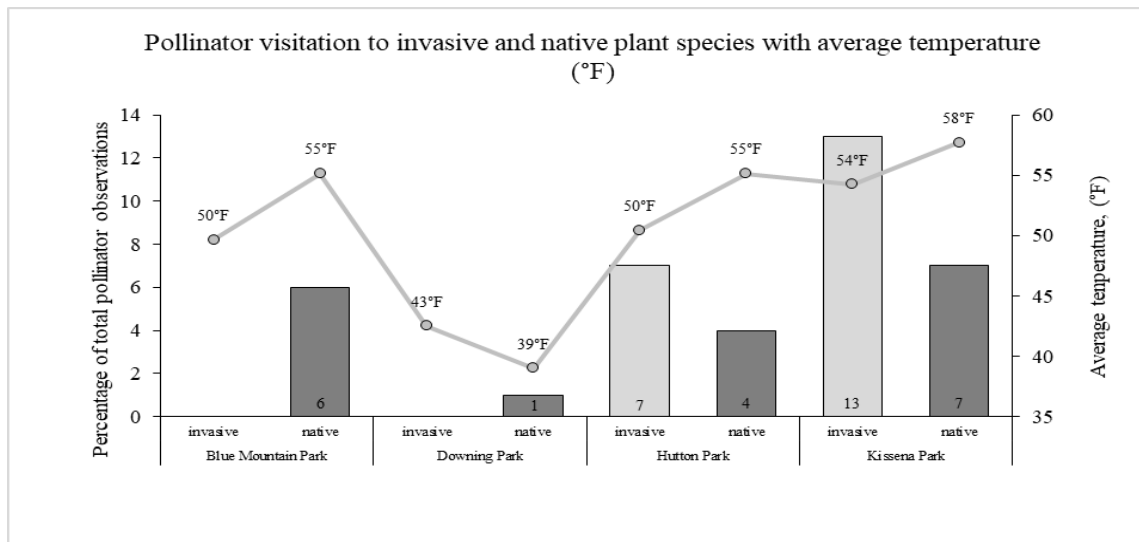


Figure 2. The goal was to see the relationship between pollinator visitation and temperature of the location; it is also to display the variation between pollinators on native vs invasive plants. The temperature during our trials varied by around 19°F and were usually taken between 10AM-12PM. If a pollinator was present it would be noted if it were on an invasive or a native plant. The colder temperatures had an impact on the presence of pollinators; Yorktown Heights (Downing Park) was the coldest and had the least pollinator activity, while Queens (Kissena Park) was the warmest and experienced the most pollinator activity.

Out of the 40 trials we observed 20 pollinator species on plants. We then took our data and converted them into percentages (%) to display on some of our graphs (Fig. 3). More pollinators were typically observed with a higher average temperature (°F); Downing Park (42°F) had the lowest average temperature and the lowest pollinator involvement as well (Fig. 2). Hutton Park (58°F) and Blue Mountain (57°F) had the highest average temperatures and the most pollinator engagement.

Pollinators at Downing Park and Blue Mountain Park had distinct preferences for native plants, the only place with more

pollinators on invasive plants was Hutton Park (Fig. 2). At Downing Park and Blue Mountain pollinators had a 100% preference for native plants, while pollinators at Kissena Park preferred invasive plants (65%) over native plants (35%) (Fig. 3). Pollinators had a slight preference for invasive plants (57%) over native plants (43%) at Hutton Park (Fig. 3).

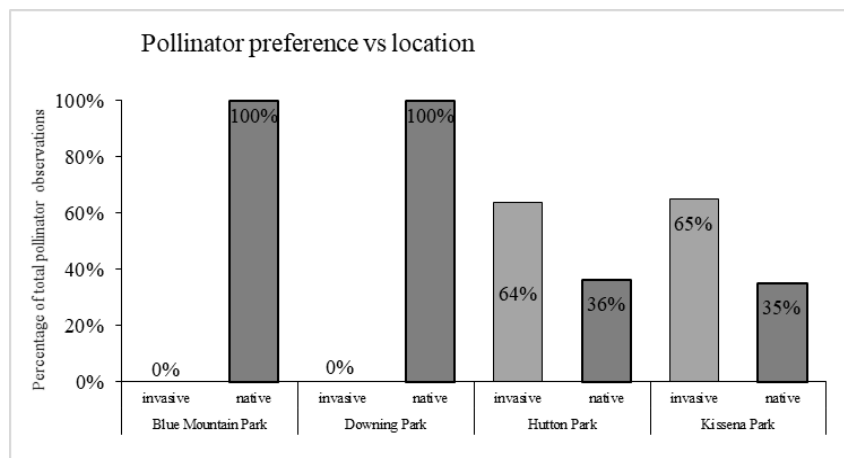


Figure 3. If a pollinator was observed, we would classify the plant based on whether it was invasive or native. 100% of the observed pollinators at Blue Mountain (Peekskill) and Downing Park (Yorktown Heights) were observed on native plants. Hutton Park (Kingston) and Kissena (Queens) both observed slightly more interactions on invasive plants (64% and 65% respectively).



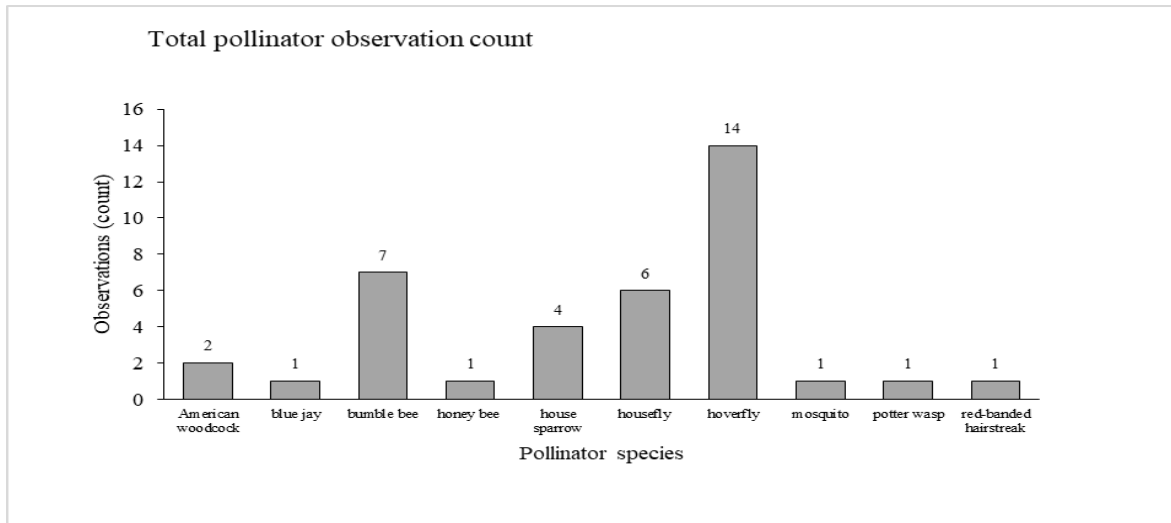


Figure 4. The graph above illustrates which specific pollinator we observed and describes how many times we saw them. This is to help illustrate which pollinators were the biggest contributors to our study. The hoverfly had the highest involvement being seen interacting with different plant species 14 times, followed by the bumble bee, which was observed 7 times.

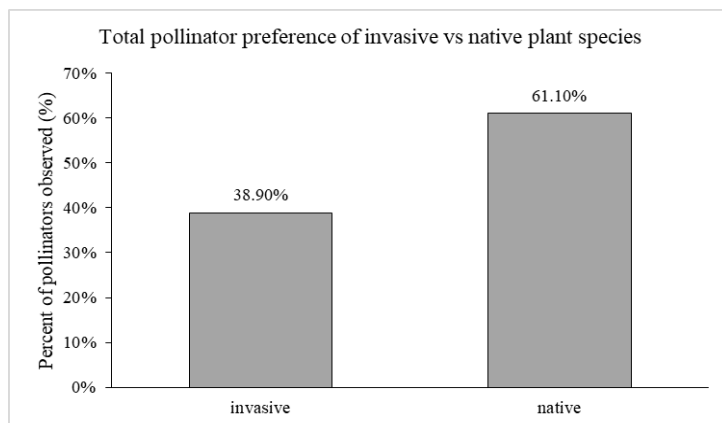


Figure 5. The ultimate goal was to see if pollinators had a preference for native or invasive plants. Out of the 40 trials we did, 20 plants had pollinators, or 50.0% of observations. The data could suggest that pollinators prefer native plants: we observed that 61.10% pollinators were on native plants, and only 38.90% were seen on invasive plants.

We also wanted to acknowledge the different pollinator species involved, so we recorded the sum of different pollinator species we saw. The hoverfly and the bumblebee offered the most pollination during our trials (Fig 4). Birds also appeared in our observations: the blue jay (*Cyanocitta cristata*) was spotted 1 time, the house sparrow (*Passer domesticus*) 2 times, and the American woodcock (*Scolopax minor*) was spotted 2 times.

It is evident from our data collection that pollinators did prefer native plants over non-native plants. In the 20 trials in which we spotted pollinators, 61.10% preferred native plants over invasive plants (Fig. 5). The plant that pollinators preferred the most in our study were asters (*Asteraceae*), accounting for 36% of the plants involved in the sightings. The sum of native plants (58%) preferred by pollinators was greater than the results for invasive plants (42%) (Fig. 6).

## DISCUSSION

The goal of this study was to determine whether pollinators prefer native or invasive plants in city parks. Our results show that pollinators in urban areas preferred native plants to invasive ones in the vicinity. This is in line with what we predicted would be seen and is reinforced by previous studies (Grass et al. 2013). Our study was conducted during the fall from mid-October to early November. As a result of the time of year, many of the non-native and native plant species were not flowering. This would virtually

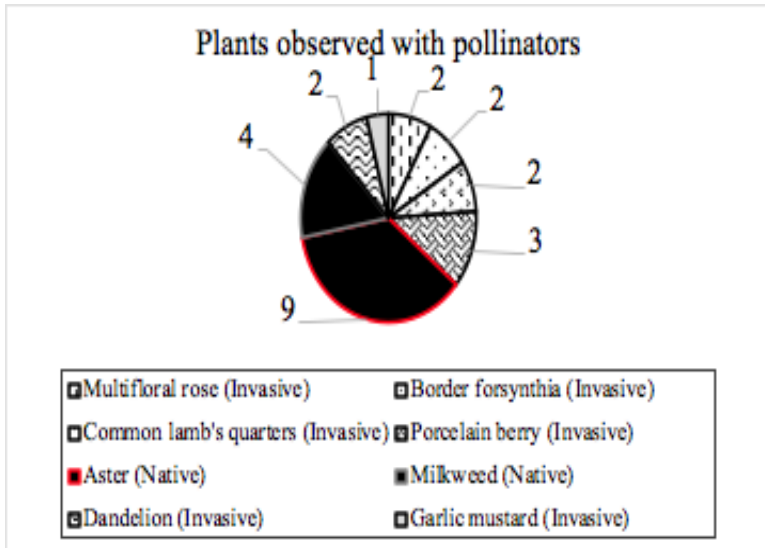


Figure 6. The pie-chart shows each individual plant species we saw a pollinator on and classifies them as native or invasive. Out of the 20 trials with pollinators we observed them on plants 25 times. The black areas of the pie-chart represent pollinators spotted on native plants. Asters (Asteraceae) had the highest number of pollinator involvement out of any other plant in the study with 9 sightings (36%). The white/gray sections of the chart represent invasive plants. We observed a greater sum of pollinators on native plants (52.63%) than on invasive plants (47.37%).

(Lavery 1992). More flowering invasive species during the data collection could have furthered the results of this study. Survival in urban ecosystems can be difficult for many plant and animal species. It has been shown that the diversity of pollinator species decreases as areas become more urbanized, so it is important to protect their interests so that vital ecosystem services are not lost (McKinney 2008). So, since this study shows that pollinators prefer native plants, the survival of those plant species should be a priority. If further studies also conclude that pollinators prefer native plants, then invasive species management plans would need to be created or altered to take this new information into account.

The density and diversity of both non-native and native plant species varied between each of the four parks visited during this study. This can make it difficult to accurately compare the sites and can skew the results. If there were flowering non-native plants at one site and no flowering non-native plants at another, then “use” of non-natives by pollinators would seem very different when it might not be in reality. In other words, the results of this study are prone to type 1 errors. A type 1 error occurs when there is a conclusion that there is a difference in pollinator preference when there is not one. Since there were also areas of the various sites surveyed that had native plants that were still flowering, it is possible that pollinator use of those plants were based on availability rather than preference. Future studies could choose specific species of non-native and native plants that are present at all the sites being surveyed.

Many of the pollinator species that were observed migrate, hibernate, or die as a result of cold winter temperatures. The threshold of too cold for these species and others that can be found in this region of New York was crossed several times over the course of this study. The temperatures during this time of year can vary greatly. There were temperatures as high as 70°F and as low as 29°F, which had an influence on the species observed and the level of activity those pollinators exhibited. During the fall months, several species of butterflies migrate, the most well-known of those being monarch butterflies. This may explain why only one species of butterfly was observed during the data collection period. When temperatures fall below 50°F, bees stop flying and return to their hives where they can keep warm and

eliminate all pollinator interactions with those plant species, since they rely on the flower parts of the plant. Porcelain berry, multiflora rose, and wild grape are just a few of the non-native species observed that are not flowering at this time. Dandelions flower from May to October and were the only invasive plant that might have been of use to any pollinators. Studies done in the future might consider changing the time of year to spring and summer months. This would ensure that there is a good chance the pollinators will have the opportunity to choose non-native plant species to pollinate. It would be an opportunity to study the effect of differences in floral presentation as well, which has been shown to influence pollinator visitation (Vanparys et al. 2008). Because invasive plants were not flowering, this study did not observe any “magnet species” effects, where plants that are highly attractive to pollinators are favored by them

survive the cold temperatures (Hogeback, n.d.). Our results show that when temperatures were higher more pollinator species were recorded. Because of the fluctuation there may have been a significant impact on pollinator activity. To add to the current understanding of plant-pollinator interactions, future studies should be done during seasons where pollinators will be more active.

In addition to recording only the species of pollinator observed, the number of individuals of each species that visited each site could have been recorded. The size of the observed areas could have also been quantified, as there were no guidelines laid out as to how large or small observed areas should have been in this study. Large areas will likely have more plants and thus more pollinators, and we do not know whether that can be reliably compared to smaller areas or areas of the same size with reduced plant density. Moragues and Traveset (2005) showed that non-native plant presence can have negative, positive, and neutral effects depending on the site, and a detailed and concise future study could show much different results than were found here.

During the data collection portion of this study a miscommunication occurred between group members resulting in inconsistencies in data to analyze. Several members selected four sites at their respective parks to go back to each week and observe for pollinators. Others only went to one site each week to observe for pollinators. One member collected data from four sites one week and then went to one site for the rest. This discrepancy in the data that was collected potentially skewed the data. Future studies might consider making sure all the data collectors collect the same amount of data by implementing stricter, more detailed guidelines for data collection during the beginning stages of the study.

## CONCLUSIONS

Based on our findings, we conclude that the pollinators in urban parks have a preference for native plants over non-native plants. Temperature was also shown to have a negative effect on pollinator abundance. Our study, which was done in cold temperatures, yielded relatively few pollinator sightings. Invasive plants have a negative effect on ecosystems and pollinators are especially at risk. Pollinators are vital for ecosystem functioning and are at risk due to increased urbanization (McKinney 2008). It is important to understand pollinator tendencies so that their populations can be preserved. Information from this study and the information gathered from future studies can be used to inform pollinator interactions with invasive plants and whether invasive plant removal and control are necessary.

## ACKNOWLEDGEMENTS

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## AUTHOR CONTRIBUTIONS

Conceptualization (all), data collection (all), data curation (all), formal analysis (all), methodology (all), project administration (all), resources (all), visualization: maps (KS, MF), visualization: figures (AJO, KS), visualization: table (AJO, KS), writing: abstract (AJO, KS, MF), writing: introduction (AJO, MF), writing: methods (KS), writing: discussion (GM), writing: conclusion (GM, KS), writing: review and editing (all).

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## APPENDIX I

Table 1. Pollinator observations on native and invasive plant species at four public park observation sites; all located in the state of New York.

Location	°F	Plant Type	Plant Species Observed	Pollinators Observed
Blue Mountain Park (Peekskill)	29°F	native	common milkweed ( <i>Asclepias syriaca</i> )	house fly ( <i>Musca domestica</i> )
	54°F	native	American aster ( <i>Symphiotrichum</i> )	hoverfly ( <i>Syrphidae</i> )
	54°F	native	swamp milkweed ( <i>Asclepias incarnata</i> )	hoverfly ( <i>Syrphidae</i> )
	65°F	native	common aster ( <i>Symphiotrichum</i> )	bumble bee ( <i>Bombus</i> )
	65°F	native	common milkweed ( <i>Asclepias syriaca</i> )	bumble bee ( <i>Bombus</i> )
	65°F	native	swamp milkweed ( <i>Asclepias incarnata</i> )	bumble bee ( <i>Bombus</i> )
Downing Park (Yorktown Hts)	39°F	native	American aster ( <i>Symphiotrichum</i> )	potter wasp ( <i>Vespidae eumeninae</i> )
Hutton Park (Kingston)	40°F	native	American aster ( <i>Symphiotrichum</i> )	house fly ( <i>Musca domestica</i> )
	40°F	invasive	border forsythia ( <i>Forsythia intermedia</i> )	American woodcock ( <i>Scolopax minor</i> ), house sparrow ( <i>Passer domesticus</i> ), blue jay ( <i>Cyanocitta cristata</i> )
	40°F	invasive	common lamb's-quarters ( <i>Chenopodium album</i> )	American woodcock ( <i>Scolopax minor</i> ), house sparrow ( <i>Passer domesticus</i> )
	52°F	native	American aster ( <i>Symphiotrichum</i> )	house fly ( <i>Musca domestica</i> ), mosquito ( <i>Culicidae</i> )
	75°F	native	American aster ( <i>Symphiotrichum</i> )	house fly ( <i>Musca domestica</i> )
	75°F	invasive	border forsythia ( <i>Forsythia intermedia</i> )	house sparrow ( <i>Passer domesticus</i> )
	75°F	invasive	common lamb's-quarters ( <i>Chenopodium album</i> )	house sparrow ( <i>Passer domesticus</i> )
Kissena Park (Queens)	47°F	native	common aster ( <i>Symphiotrichum</i> )	hoverfly ( <i>Syrphidae</i> )
	47°F	invasive	porcelain berry ( <i>Ampelopsis brevipedunculata</i> ), multiflora rose ( <i>Rosa multiflora</i> ), dandelion ( <i>Taraxacum</i> )	hoverfly ( <i>Syrphidae</i> )
	49°F	native	common aster ( <i>Symphiotrichum</i> )	bumble bee ( <i>Bombus</i> ), hoverfly ( <i>Syrphidae</i> )
	49°F	invasive	porcelain berry ( <i>Ampelopsis brevipedunculata</i> ), garlic mustard ( <i>Alliaria petiolata</i> )	house fly ( <i>Musca domestica</i> ), hoverfly ( <i>Syrphidae</i> )
	49°F	invasive	porcelain berry ( <i>Ampelopsis brevipedunculata</i> ), multiflora rose ( <i>Rosa multiflora</i> ), dandelion ( <i>Taraxacum</i> )	bumble bee ( <i>Bombus</i> ), hoverfly ( <i>Syrphidae</i> )
	70°F	native	common aster ( <i>Symphiotrichum</i> )	honey bee ( <i>Apis mellifera</i> ), red-banded hairstreak ( <i>Calycopis cecrops</i> ), hoverfly ( <i>Syrphidae</i> )

**\*All observation sites with asters (*Asteraceae*) observed at least one pollinator at site.**

# BLUE JAYS ARE MORE LIKELY TO SPEND TIME FEEDING IN COVERED AREAS

Ryan A. Ramirez, Andrew Bowman, Sammy Jimenez, and Eric Beilin

## ABSTRACT

*In response to urbanization, many animals have evolved certain behavioral syndromes to cope. The most common behavioral traits that maximize fitness and survival, in terms of optimal foraging theory, is boldness vs foraging in covered sites. To study this, we observed blue jays foraging on peanuts in covered and open areas. We found that blue jays, on average forage, for a longer period of time in covered areas than in exposed areas. Therefore, our data backs the idea that certain behavioral syndromes in animals towards urbanization can greatly increase their chances of efficiency in foraging, compared to others. Furthermore, our results are important to guide future research on foraging behavior in human-dominated landscapes to encourage better policies on ecological management.*

Keywords: Behavior, biological fitness, blue jays, covered and exposed areas, optimal foraging, urbanization

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## INTRODUCTION

Optimal foraging theory was developed to explain how certain species hunt or scavenge for food sources in accordance with their environment (Sih and Christensen 2001). Many factors that play a role in an animal's foraging patterns include, but are not limited to, the type of food source, the availability of it, the geography of its foraging areas, the scavenger's physical limitations, predator abundance, and competitor abundance (even among its own species). For example, some animals, especially those that are easy prey, like rodents, are concerned with the balance of risk and reward during their foraging activities. By considering the factors mentioned, particularly the type of location they forage in, the animals are able to recognize if the reward outweighs the risk (Sih and Christensen 2001).

Unfortunately, outside intervention can drastically distort a species' optimal foraging in regards to location, food preference and availability. The most common disruption is human urbanization: Replacing natural habitats with human landscapes such as buildings, roads, and houses (Belinsky et al. 2019). While urbanization is a recreational way of improving human life, as many population densities increased, so did the exploitation of many natural resources, including water, trees, plants, and fertile lands for crops. This leads to rapid and dramatic reductions in abundance and diversity of wildlife, with them either dying or pushed out of their natural environments (Tryjanowski et al. 2016). However, some species of animals have evolved certain behavioral syndromes that allow them to cope with urban environments, especially when their foraging grounds are within them (Tryjanowski et al. 2016).

Despite urban environments being superabundant in various food sources, human presence can serve as a predation risk for some animals. Certain animals cope with this through boldness [neophilia] or avoidance against something novel [neophobia] (Tryjanowski et al. 2016). The greater the frequency of exposure, the greater the boldness. Also, regarding predation as a risk, some animals compensate for this by feeding in areas that offer protective cover. In other scenarios, many prefer to forage in areas closer to where they live (Sih and Christensen 2001). Unfortunately, across many groups of animals, not all of their members share one coping mechanism to urbanization. On the other hand, the group may evolve a behavioral mechanism that would risk the safety and stability of the group. Therefore, optimal foraging theory is not always recognized among groups due to unresolved dissent or intraspecific competition (Jones et al. 2019). For example, if there is decreased diversity of food resources in urban environments, the foraging patterns would become more predictable, making them easier to track. These patterns include group size, foraging duration, and distribution of the foraging events (Jones et al. 2019). Furthermore, if some members within a group are more risk-prone and neophilic compared to the rest, this may incur a fight for dominance to influence the overall group social structure and behavior (Jones et al. 2019).

The most common animals used to observe optimal foraging theory are birds, given that they are numerous in various habitats, including urban, and that they have faster response times to disruptions thanks to flight (Belinsky et al. 2019). In this case, optimal foraging theory assumes that the most fit bird would be the most efficient feeder, if they can overcome the risk of predation. Therefore, natural selection would favor that individual over all the others because they cannot adopt those same behaviors (Werner and Mittelbach 1981). We observed how often blue jays (*Cyanocitta cristata*) visited the food source and how long they stayed at the food source. Furthermore, we hypothesized that blue jays were more likely to visit and stay longer at a covered feeder rather than a feeder in an exposed area.

## METHODS

We used one makeshift bird feeder made out of cardboard. The food source we decided on was 50 peanuts (*Arachis hypogaea*). Two sites were set up in the front yard of a house. The exposed site was in the center while the covered site was in a bush. Also, the ground of the covered site was filled with leaves that had fallen from trees. For the exposed site, two trials were recorded, showing every blue jay's appearance and duration at the feeder. For the covered site, two trials were recorded for the same data. After every trial, we refilled the feeder up to 50 peanuts.

For each trial, we recorded the length of time every individual blue jay spent each time it visited the feeder. We used this data to calculate the average number of seconds, per visit for every feeder. We then calculated the standard deviation and standard error for each trial. Lastly, we combined the data for both covered trials and exposed trials in order to evaluate the differences in average-visit-duration and total number of visits.

## RESULTS

Our results show that blue jays tend to spend more time feeding in covered locations than exposed locations (Fig. 1). We found that in the two trials using covered feeders, blue jays spent on average 13.24 seconds each time they came to the feeder. In the uncovered trials, jays spent only 9.36 seconds at each visit. This distinction can be seen across all four trials. Trial 3 had the highest visit duration of both covered trials, and was still 1.29 seconds below Trial 4, which was the lowest duration in covered trials (Fig. 2). This difference is not due to less visits being made to covered feeders. We recorded a total of 96 visits to covered feeders, representing 56.1% of total visits recorded (Fig. 3). We also recorded the distribution of times blue jays spent feeding. The majority of results are clustered under the 15 second mark (Fig. 4). Since blue jays did not seem to spend time at the feeder together, more research to determine if this timing is an effect of competition of optimal foraging theory is needed.



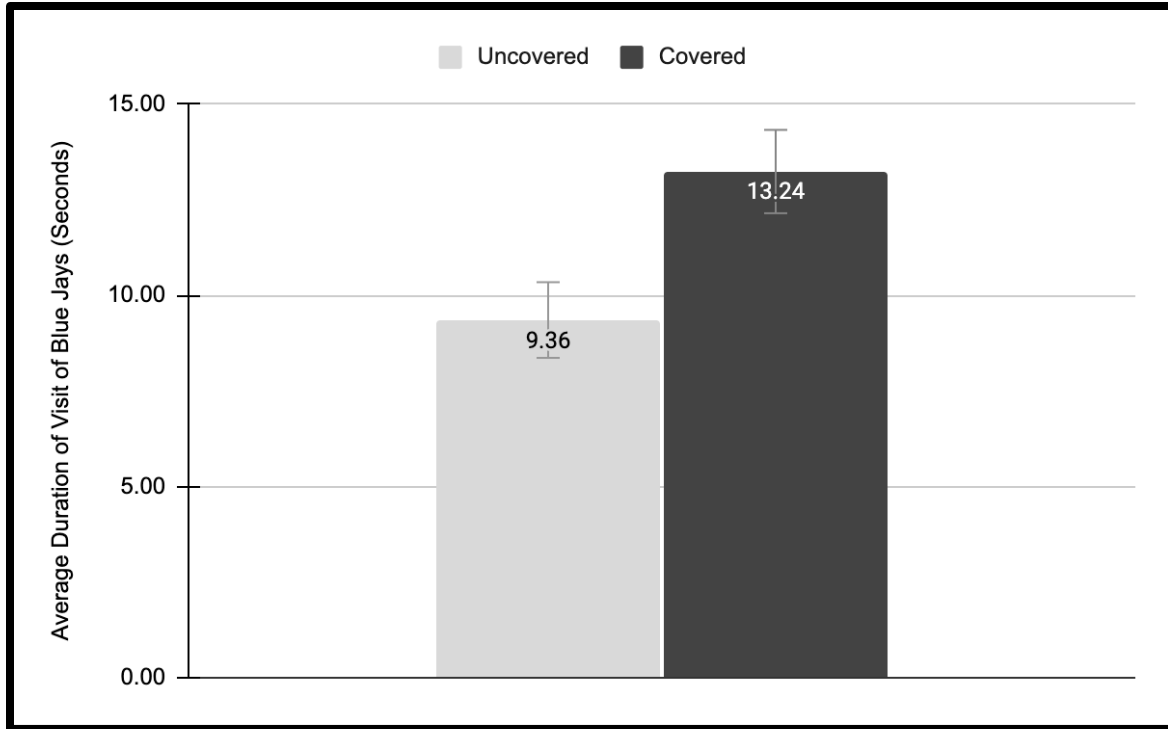


Figure 1. Blue Jays tend to stay for longer at the covered feeders than the feeders left in the open. Error bars represent standard error.

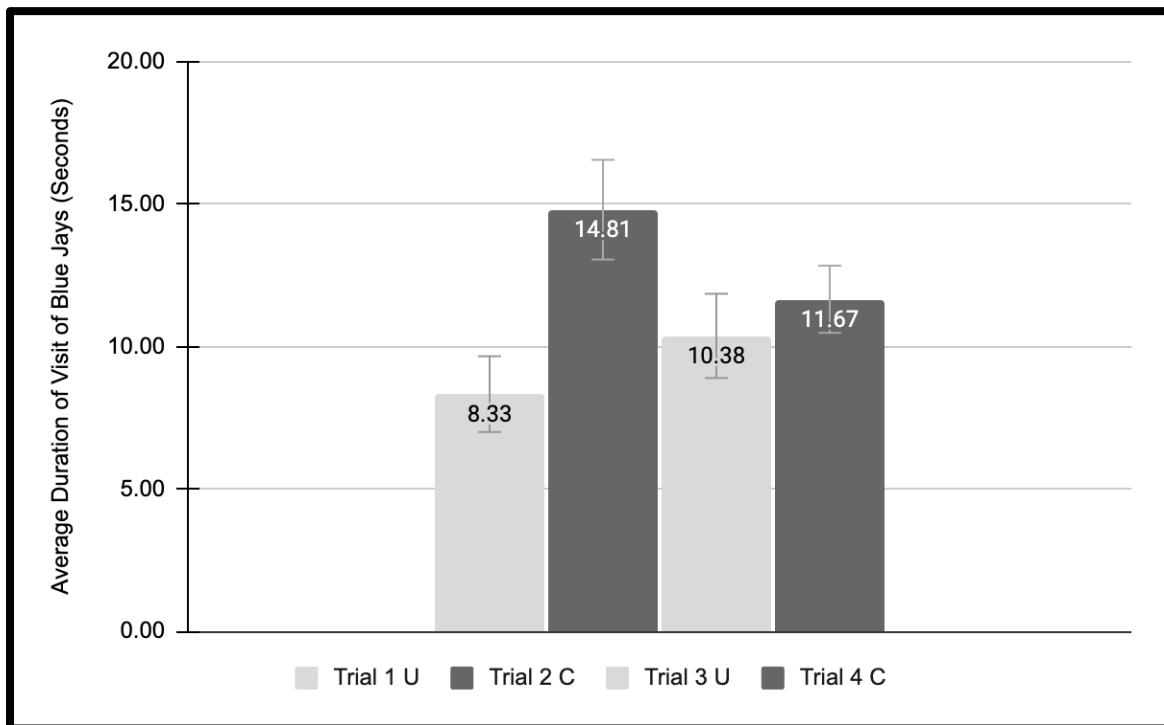


Figure 2. The average visit duration from all four trials. The trials with the highest average duration (Trial 2 and Trial 4) were both performed with a covered feeder. Error bars represent standard error.

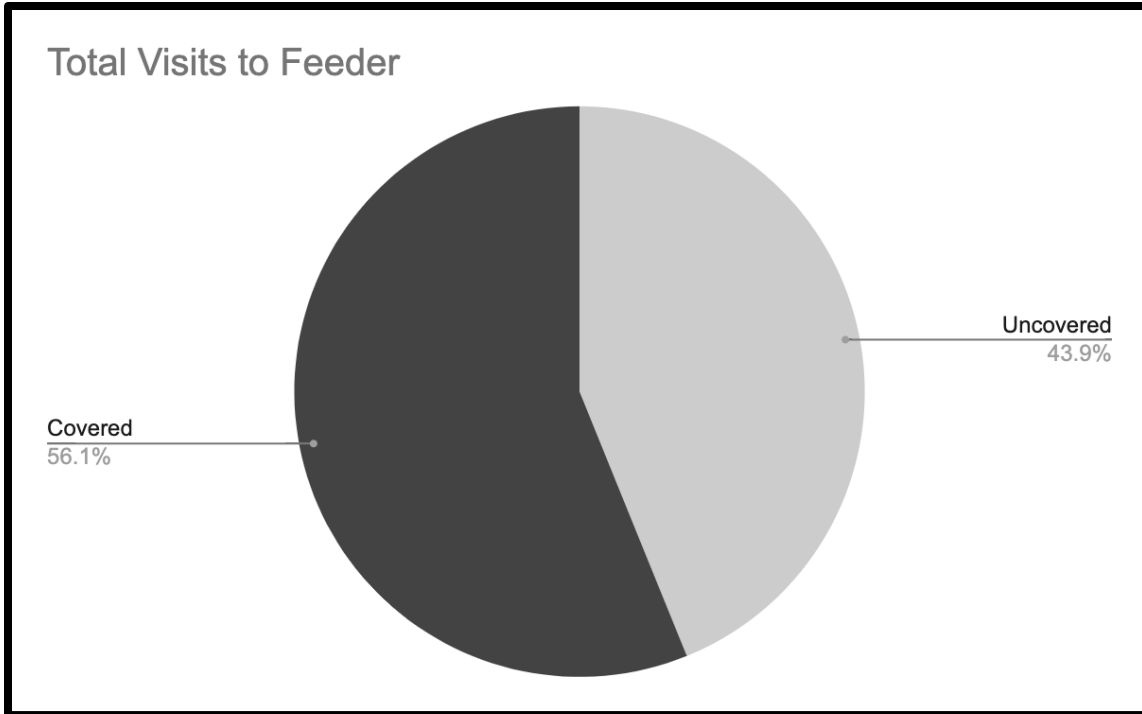


Figure 3. When all results were recorded, we saw more total visits to the covered feeders than to the uncovered feeders.

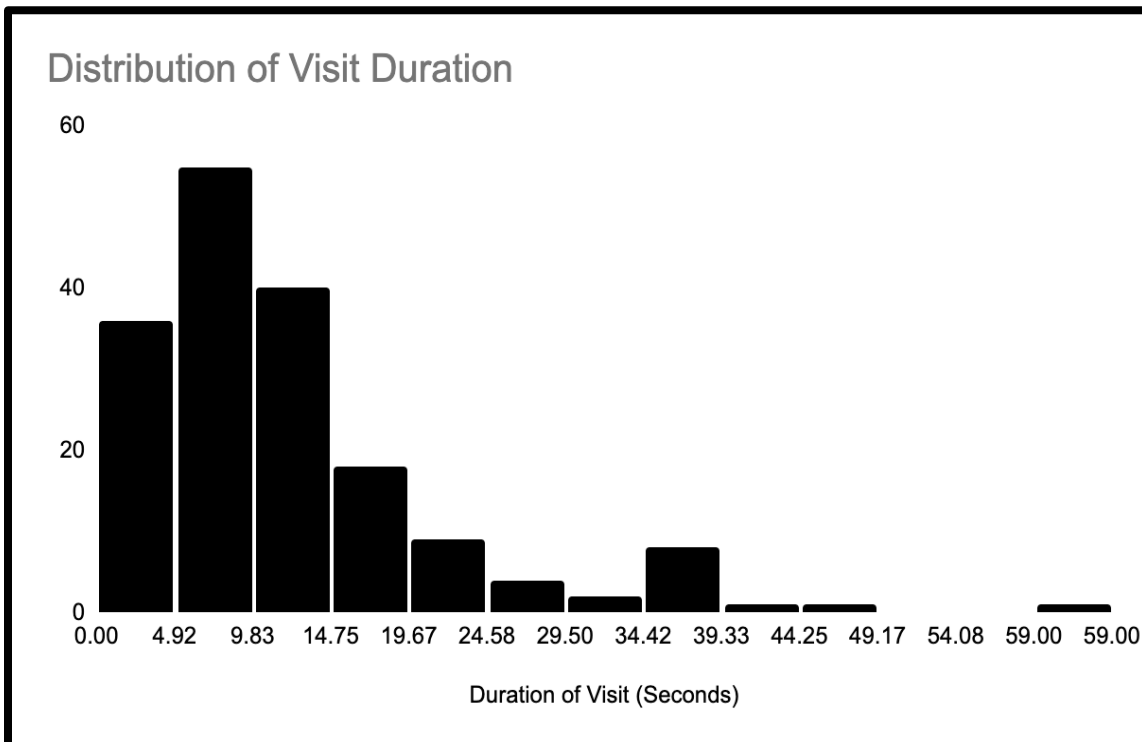


Figure 4. The distribution of the duration of time spent at the feeder per visit.

## DISCUSSION

Our results are consistent with our hypothesis that blue jays were more likely to visit and remain for longer at a feeder with cover, rather than a feeder in an exposed area. Our analysis shows that on average, blue jays stayed for longer in the covered feeders than the uncovered feeders by 3.88 seconds. Additionally, the covered feeders were visited by blue jays 96 times compared to 75 visits to the uncovered feeders. This is consistent with optimal foraging theory, which posits that organisms will cease to forage at a patch when the benefit of foraging is equal to the combined metabolic cost and perceived risk of predation (Martin 1985, Pyke 1978). Some of these determinants are the energy densities of food sources, the abundance of those food sources, the geography of the area they forage, carrying limitations, predators and competition with others all play into optimal foraging theory (Wilson 1976). Since the rate of searching, along with the type and density of the food sources are all equal, we can determine that the perceived risk of predation is the reason for any variance in foraging behavior.

These results are consistent with existing research on optimal foraging theory. Analysis of the giving up density in birds in the greater Phoenix area found that birds will cease going to a patch significantly sooner in open areas, noting that this disparity was lesser in the urban area than the surrounding desert (Schochat et al. 2004). Further research is needed to determine if this urbanization difference is maintained in other habitat types. This difference has also been found when multiple artificial patches were presented at once, as an analysis of white-throated sparrows (*Zonotrichia albicollis*) shows that the birds will nearly deplete a site located closer to cover before moving away (Schneider 1984). Thus, the optimal foraging site for is the intersection between density of food and distance from cover. Schneider (1984) found a key interaction between optimal foraging theory and dominance: That subordinate birds will be forced to feed farther from cover in order to acquire enough food. This matches up with our qualitative observations that blue jays appeared to be chasing other jays away from the covered feeder.

Our analysis of optimal foraging theory in blue jays does not answer how the birds perceive risk and reward. The original optimal foraging theory assumed animals that were perfectly knowledgeable of their environment, and based predictions off of this. Research has shown that this is not the best predictor for foraging behavior. Recent studies have shown that animals incrementally update their decision making based on recent experiences (Marshall et al. 2013). More research is needed on this to determine the weight animals place on experiences at foraging patches. This is especially critical in a world where human influences can alter environments drastically in a short period of time. Studies have already shown that current models cannot perfectly predict foraging behavior (Killeen et al. 1996). Further research into this topic would improve our understanding of how animals interact with their environment, which is critical to protecting biodiversity.

When foraging, animals may repeatedly return to areas they foraged in. In the instance of a feeder, it seems like the birds feeding from it are repeat customers to the spot rather than them all being different birds (Jones et al. 2019). Further testing could be done by tracking the animals that come to the feeder to see if they have a pattern of going back and forth: To an area where they know there is food versus aimlessly searching for food in an unknown area. Further research could be done to examine the changes to foraging patterns when birds are breeding because it was discovered that they are stricter in their food selection when raising young (Sauter et al. 2006). Offering a variety of food choices could also result in changes in behavior. Blue jays are known to prefer certain types of nuts and seeds, and this preference could be enough to overcome the risk of exposed feeders (Moore and Swihart 2006).

There are several improvements we would consider for our study. During data collection, if we were to increase the sample size by taking longer videos or more videos, that would result in a more accurate model. Another way to increase sample size would be to have more locations under surveillance, such as four covered areas and four uncovered to see the effects across a broader geographic range. To take this even further, tests could have been done during each season, as was done in a study by Lewis (1982), where he had three time periods encompassing all the seasons. Studies have shown that animals have differing methods of foraging depending on shifts in their environment conditions, such as change in

season (Kuwae et al. 2010). Another improvement would be to have cameras placed at more angles around the feeder, to clearly determine the length of time a blue jay was in close proximity to the feeder.

## CONCLUSION

The way animals forage has been greatly affected due to human intervention. So much space is now dominated by human activities and infrastructure, that many animals now live within the sphere of human influence. This has resulted in altered foraging behavior, particularly when comparing urban and wild populations (Schochat et al. 2004). An example is that many animals regularly scavenge from human-food waste sources, such as garbage cans (Sauter 2006). With the abundance of animals living in human dominated habitats, studying their behavior can help us understand the impact we have on natural environments and inform ecological guidelines. Further studying can even help us comprehend the reasons behind animals' foraging patterns and contribute to a better understanding of the natural world as a whole.

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## AUTHOR CONTRIBUTIONS

Conceptualization - all. Data collection - RR. Data curation - EB and SJ. Formal analysis - EB and SJ. Methodology - all. Project Submissions - RR. Resources - all. Visualization: Figures - EB. Writing: Intro - RR and SJ. Writing: Methods - RR. Writing: Results - EB. Writing: Discussion - EB and AB. Writing: Conclusion - AB. Writing: Abstract - RR. Writing: Acknowledgements - RR. Writing: Literatures Cited - all. Writing: Reviewing and Editing - EB and RR.

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