NEAR, FAR, WHEREVER YOU ARE...SPECIES I.D. IS HARD: COMPARING DRONE- AND FIELD-BASED ASSESSMENT OF TWO INVASIVE PLANT SPECIES

by

Shaina G. Thompson

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Shaina G. Thompson

has been approved for

The Evergreen State College

by

Kevin Francis, Ph. D.

Member of Faculty

Date

ABSTRACT

Near, Far, Wherever You Are...Species I.D. Is Hard: Comparing Drone- and Field-Based Assessment of Two Invasive Plant Species

Shaina G. Thompson

Restoration of degraded land requires management of invasive species to allow for the regeneration of native plants, but resources are extremely limited (Sheley & Smith, 2012). Mapping abundance of invasives across restoration sites provides land managers with a critical tool for efficient planning, management, and monitoring (Yager & Smith, 2009). Remote sensing technology has greatly improved capture of higher resolution imagery across time, space, and the light spectrum, coupled with increased computing power enabling faster data processing (Neyns & Canters, 2022). There are many exciting applications for this technology, and in the context of environmental work, there is great potential to improve classification of vegetation (Neyns & Canters, 2022), which will support monitoring of ecosystem health and habitat restoration. Once collected, whether in the field or remotely, this data can be collated in a geographic information system (GIS) to create a map which allows users to measure species abundance, and track changes over time (Yager & Smith, 2009). This study assesses how accurate manual interpretation of drone imagery enables measurement and geospatial analysis of invasive plant species across mixed habitat types. Specifically applying these mapping techniques when measuring the distribution of Himalayan blackberry (Rubus armeniacus) and reed canary grass (Phalaris arundinacea) at a forested wetland restoration site. On average, drone data only intersected field data by 37%. Overall, manual interpretation of drone imagery resulted in an overestimation of reed canary grass, and underestimation of Himalayan blackberry compared to data collected in the field.

Table of Contents

List of Figures vi
List of Tables vii
Acknowledgements viii
1. Introduction 1
1.1. Invasive Species Management in Ecological Restoration 1
1.2. Applying remote sensing technology to managing invasives within habitat restoration projects 3
1.3. Cost-efficient mapping enables greater efficiency of managing invasives across various habitats 4
1.4. Significance 5
1.5. Roadmap 7
2. Literature Review 8
2.1. Introduction 8
2.2. Managing invasive plants within the Wildland-Urban Interface 9
2.2.1. Challenges for invasive plants management 14
2.2.2. Study species 15
2.3. Mapping 21
2.3.1. Technologies 22
2.3.2. Mapping approaches 26
2.3.3. Monitoring 27
2.4. Study Site 28
2.4.1. Historical land use and current status 32
2.4.2. Restoration goals 33
2.5. Conclusion 34
3. Methods 35
3.2. Drone imagery orthomosaics acquisition 36
3.3. Manual interpretation of the drone imagery 37
3.4. Field data collection and Field Maps schema 39
3.5. Comparing data collection methods 44
4. Results 47
5. Discussion 53
5.1. Assessment of methods 53
5.2. Method efficiency 53

5.3. Accuracy limitations of field data 54
6. Conclusion 55
References 58

List of Figures

Figure 1. Housing in the Wildland-Urban Interface from 1990-2020 10

Figure 2. Map of WUI in Washington State 11

Figure 3. Annual Housing Units Authorized in the United States.. 12

Figure 4. Reed canary grass at various stages of inflorescence 18

Figure 5. Remote Sensing Publications in Invasive Plants Research 24

Figure 6. Map of WRIA 13 Deschutes Watershed 30

Figure 7. Layer of Drone Polygons... 39

Figure 8. Various Screenshots of ESRI’s Field Maps Application 40

Figure 9. Screenshot of Data Collection Map on Field Maps.. 42

Figure 10. Layer of Field Polygons 43

Figure 11. Layer of Intersect Polygons... 45

Figure 12. Study Area Impacted by 2023 Restoration Work... 48

Figure 13. Subsection of Study Site Used for Data Analysis 49

Figure 14. “Summary Statistics” Geoprocessing of “Area_sqm” Fields..................................... 50

Figure 15. Intersect Polygons with Labels... 51

List of Tables

Table 1. Long-term Control Methods for Himalayan Blackberry 17

Table 2. Recommended Best Practices for Managing Reed Canarygrass in the PNW 20

Table 3. Sensors Used in Vegetation Classification Studies 26

Table 4. Summary of Statistics of Drone and Field Polygons by Species.................................... 52

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1. Introduction

1.1. Invasive Species Management in Ecological Restoration

Most species are unaware of private property lines, fences, and other borders drawn by humans, unless demarcated by uninhabitable barriers such as concrete or asphalt. Plants, however grounded they seem to be, can travel great distances. Seeds floating in the air or in streams, traveling on animals, especially humans, dispersion can occur without the intention or awareness of the carrier (Beckman & Sullivan, 2023). Root fragments or pieces of the plant capable of propagation may be carried far from the original plant, in the fur of animals or treads on the soles of hiking boots (Soll, 2004). In some cases, chosen plants are brought to new habitats because they are beneficial to those planting them. National big box stores such as Home Depot sell seeds and plants with no regard to their native habitats. Private citizens may not be aware, or do not care, about invasive species management, which means governments must efficiently invest in public education, policy interventions, and efficient information networks (Wallace et al., 2020). Land managers engaged in ecological restoration have the difficult task of managing invasive species.

Restoration of disturbed or degraded land usually requires management of invasive plant species to allow for the regeneration or re-establishment of native plants (Sheley & Smith, 2012). Invasion ecology is rife with theories, values, and vocabulary which highlight varied, and often conflicting, perspectives on how humans manage other species to maintain “wild” spaces or restore landscapes altered by human activities (Sax et al., 2023, Simberloff et al., 2013). The word “invasion” is loaded with negative connotations, conjuring images of armies or colonizers taking over new lands. Some researchers call for acknowledgment of this bias and recognition of the ecosystem services provided by non-native species as part of reevaluating predominant academic paradigms of studying them as inherently damaging (Sax et al., 2023). That being said, within invasion science research, non-native or “alien species” can be categorized along a naturalization-invasion continuum. At one end are “casual” or ephemeral members of a species, followed by the naturalized, and then invasive, where reproduction and establishment has spread far beyond the place of introduction (Richardson et al., 2011). In order to preserve, conserve, or restore natural habitats, a majority of the scientific community prioritizes native species, and in some cases, advises complete eradication of non-native invasive species.

Invasive species management calls for active monitoring. This monitoring is important for understanding scope of work to limit spread, focus efforts, and assess effectiveness of different management techniques over time. Invasive species monitoring can be expensive and habitat restoration projects are generally constrained by limited resources. Mapping abundance of invasives by geolocating their presence across restoration sites provides restoration project managers with a critical tool for efficient planning, management, and monitoring treatment of invasive plants (Yager & Smith, 2009). Hence, there is substantial interest in how remote sensing, whether by satellite, aircraft, or drones, can help monitor the spread of various species across spatial and temporal scales. Here are some promising directions: coupling these remote sensing platforms with machine learning to automate image classification (Neyns & Canters, 2022), testing these opensource algorithms on new environments and species (Dash et al., 2019), and human-classification or manual interpretation, which may be as effective at identifying certain species, depending on timing, phenology, and available resolution (Dash et al., 2019, Hill et al., 2017). At this time, field sampling is required to train classification models and validate the accuracy of remote sensing research (Dash et al., 2019). Moving from research to application, land managers interested in mapping may be able to meet certain goals through the remote sensing of invasive plant species.

1.2. Applying remote sensing technology to managing invasives within habitat restoration projects

Remote sensing technology provides promising avenues for unprecedented levels of data collection due to improvements in capturing higher resolution imagery across time, space, and the light spectrum (Neyns & Canters, 2022). There are three primary ways of collecting remotely sensed data: satellies, manned aircraft, and unmanned aerial vehicles (UAV), which are also referred to as unmanned aircraft systems (UAS) or drones. Major advancement in sensors capturing data beyond the visible light spectrum has opened up new possibilities in differentiating elevation based on reflected light, but this study will not address multispectral imaging or Light Detection And Ranging (LiDAR). For more on those topics, see Neyns & Canters (2022), Rajah et al. (2019), Chance et al. (2016), and Narumalani et al. (2009).

Another area of technological advancement enabling the growing application of remote sensing is increased computing power which enables faster data processing and analysis (Neyns & Canters, 2022). Researchers of remote sensing are testing the boundaries of these capabilities by applying machine learning, or training computers with verified data to correctly identify target species or other ground cover types (Neyns & Canter, 2022). In its simplest form, analysis of imagery consists of differentiating and labeling pixels or objects, and is referred to as classification (Dash et al., 2019). There are many exciting applications for this technology, and in the context of environmental work, there is great potential to improve the classification of vegetation across various landscapes, and even urban areas (Neyns & Canters, 2022). Alternatively, manual interpretation of remotely sensed imagery can be an effective method of analysis in the context of invasive species monitoring (Bradley, 2014).

Leveraging machine learning for detecting invasive species can support the breadth and capacity for monitoring of ecosystem health, once accuracy of a classification model has been reasonably established. Classification models must be paired with an algorithm to automate the process, and several open-source algorithms have been developed. One such algorithm, Random Forest, is most used due to its flexibility and high accuracy across multiple applications (Dash et al., 2019). Once collected, whether in the field or remotely, this data can be collated in a geographic information system (GIS) to create a map which allows users to measure species abundance and track changes over time (Yager & Smith, 2009). However, when feasible, collecting data from the ground is the most accurate data collection method, but potentially the most expensive and time-consuming (Mack et al., 2007). Field sampling is an integral part of remote sensing research to verify the method's error rate (Dash et al., 2019). By testing different remote sensors on various habitats and species, research can inform land managers on the decision to invest in remote sensing or data collection on the ground.

1.3. Cost-efficient mapping enables greater efficiency of managing invasives across various habitats

The practical framework of my research problem revolves around restoration work and efficiently managing invasive plant species within the confines of extremely limited resources across diverse habitats. Drones might offer a viable way to assess and monitor the presence of certain species across a variety of habitat types, which may save time and money (Neyns & Canters, 2022). Drone-mounted sensors can achieve very high resolutions and this method is generally less expensive than manned aircraft, enabling data acquisition at more flexible spatial and temporal scales (Dash et al., 2019, Feng & Gong, 2015). And while automated classification is not yet reliable in complex habitats, manual image interpretation provides an alternative approach (Dash et al., 2019, Bradley, 2014). However, if analyzing drone imagery does not enable accurate assessment and measurement of the spread of invasives, whether it be based on the researcher's limitations, technological shortcomings, or site conditions, it would be more efficient to build maps from data collected during field work. Having a clearer understanding of remote sensing capabilities in this application can inform which step in the invasive species management process data collection would be most efficient (Chance et al, 2016). Therefore, my general research question is: **How well does manual interpretation of drone imagery enable accurate measurement and geospatial analysis of these species across mixed habitat types, compared to field surveys?**

To answer this question, I focused on a particular site where restoration managers are trying to control and monitor invasive species, especially Himalayan blackberry (*Rubus armeniacus*) and reed canary grass (*Phalaris arundinacea*), as efficiently and effectively as possible. In addition to answering broader research, I wanted to help produce a geodatabase for Olympia Coalition for Ecosystem Preservation (OlyEcosystems) to support their management and restoration goals. For many year, their newest property, now called Deschutes River Preserve (DRP) was a dairy farm. The preserve is over 300 acres of mixed habitats located in a suburban area southeast of Olympia, Washington. Restoration of previously grazed areas and agricultural lands within the preserve requires planting of native species, which in turn requires proactive invasives management (OlyEcosystems, n.d.). This is the basis for my site-specific research question: **How do these mapping techniques compare when measuring the distribution of Himalayan blackberry (*Rubus armeniacus*) and reed canary grass (*Phalaris arundinacea*) at the Deschutes River Preserve?**

1.4. Significance

Ecosystems are incredibly complex, providing the foundational elements of life for billions of species on this planet. In recent history, the pace of change driven by human development had deleterious effects on the environment (Mirzabaev & Wuepper, 2023). The current rate of land degradation is 10 times higher than that of restoration (Le et al., 2016).

Economists who have attempted to assign monetary values to ecosystems estimate global ecosystem degradation costs anywhere from $300 billion USD (Nkonya et al., 2016) to $6 trillion every year (Sutton et al., 2016), depending on the methodology used (Mirzabaev & Wuepper, 2023). When faced with these figures, it becomes clear this trend must be reversed, for both existential and economic reasons.

This study presents a direct comparison of data collected through drone imagery and field surveys of two invasive species across several different habitat types. By comparing the data collected through different methods, this study will examine the accuracy of using remotely sensed data in the context of managing these invasive species on a restoration site with mixed habitats. Field surveys are time consuming and expensive – this study explores whether drone imagery can provide “accurate enough” identification of two prolific invasives, Himalayan blackberry (*Rubus armeniacus*) and reed canary grass (*Phalaris arundinacea*) across a variety of habitat types including grasslands, wetlands, forest, and developed areas. In this study, “accurate enough” refers to the presence of either invasive species within a half acre, providing a large margin of error, or in geographic terms, buffer. In conversation with the land manager, identification and geolocation were higher priorities over precise measurement of the study species.

These species are highly visible in the summer, and despite seasonal senescence, are still identifiable in the winter, which is why these specific species were selected for this study. After analyzing the drone imagery, I conducted field surveys to map out patches of these species by dropping points along their perimeter. I used an app designed by the Environmental Systems Research Institute (ESRI) to enable streamlined data collection and analysis. The resulting field survey polygons were then compared to the initial polygon layer based on the drone imagery analysis. There are several geoprocessing tools in ArcGIS software which make a great many types of data analysis possible. For this case study, the Intersect tool will produce a new polygon layer of where these polygons overlap, enabling numerical outputs such as percentage of area identified correctly by drone predictions, and total acres of coverage of each species within the assessed range (ESRI, n.d.). The workflow I have designed will enable restoration managers to more effectively budget, coordinate, and apply treatments, as well as measure outcomes over time with subsequent mapping, using smartphones and access to GIS software.

1.5. Roadmap

The literature review provides context for this research by describing invasives management within the Wildland-Urban Interface (WUI), challenges land managers face, as well as an overview of the study species and methods for their removal. The next section of the literature review outlines some mapping technologies and approaches, providing insight into the geospatial analysis that can be used by land managers engaged in restoration work to support their invasives management. The final section of the literature review orients this research to the study site, providing historical and current land use, as well as the restoration goals for the preserve.

Creating usable products at each step of the way, I needed a way to differentiate between methods and results. My approach has been to focus on the steps and products that are transferable to other sites in the Methods chapter. And in my Results, I describe what is specific to this site and study. I used examples from this site in my Methods chapter to illustrate how layers could be designed. The discussion section describes the differences and overlaps between the drone imagery map and field survey map. This discussion also highlights the costs and benefits of each data collection method, which can inform decisions around mapping for land managers of smaller but highly diversified habitats.

2. Literature Review

2.1. Introduction

This literature review examines current scholarship on restoration of degraded lands due to human-caused disturbance, specifically the practice of managing what are commonly referred to as invasive plant species. The first section provides context for why this type of habitat restoration in disturbed areas within the wildland-urban interface (WUI) is urgent in the face of climate change. This review will also highlight how invasives management is fraught with biological complications, resource constraints, and competing priorities. We must apply cost-benefit analysis to practices, without overlooking their sustainability and overall ecosystem health. A brief overview of the two invasive species being studied, Himalayan blackberry (*Rubus armeniacus*) and reed canary grass (*Phalaris arundinacea*), will highlight the potential management strategies that could be implemented.

The next section of this literature review will focus on mapping. Potential cost-savings may be found in the form of remote sensing and geospatial technology, and many studies have examined different applications of satellite, aerial, and unmanned aerial (drone) imagery for assessing land cover, forest types, and coarse vegetation (Neyns & Canters, 2022, Dash et al., 2019, Feng et al., 2015, Iovan et al., 2008, Mathieu et al., 2007, Walter, 2004). As this technology continues to evolve and becomes accessible at lower price points, land managers may be able to leverage imagery to enrich existing or create new geodatabases. There is a growing capability to accurately assess the abundance and spread of certain vegetation beyond tree species, quickly and over time, at lower costs (Chance et al., 2016, Bernthal & Willis, 2004). This research will focus on drone imagery, as it is the option which balances very high resolution and quality with cost and flexibility.

The third and final section of this literature review will provide a thorough site description for this research. This study will examine the Deschutes River Preserve (DRP), using its historical land use, status, and restoration progress as a case study for comparing invasives monitoring with drones and field surveys. This site is situated along the Deschutes River, a watershed that connects the forests of the Cascades to the Puget Sound (OlyEcosystems, 2023). Part of a patchwork of parcels that have been acquired by the Capital Land Trust and OlyEcosystems, these pockets of preservation provide critical habitat interspersed throughout housing and other developed areas. The final section of this literature review will briefly outline the challenges and restoration goals of DRP, providing the foundation for this study of utilizing drone imagery to assess invasive species in a suburban, mixed habitat.

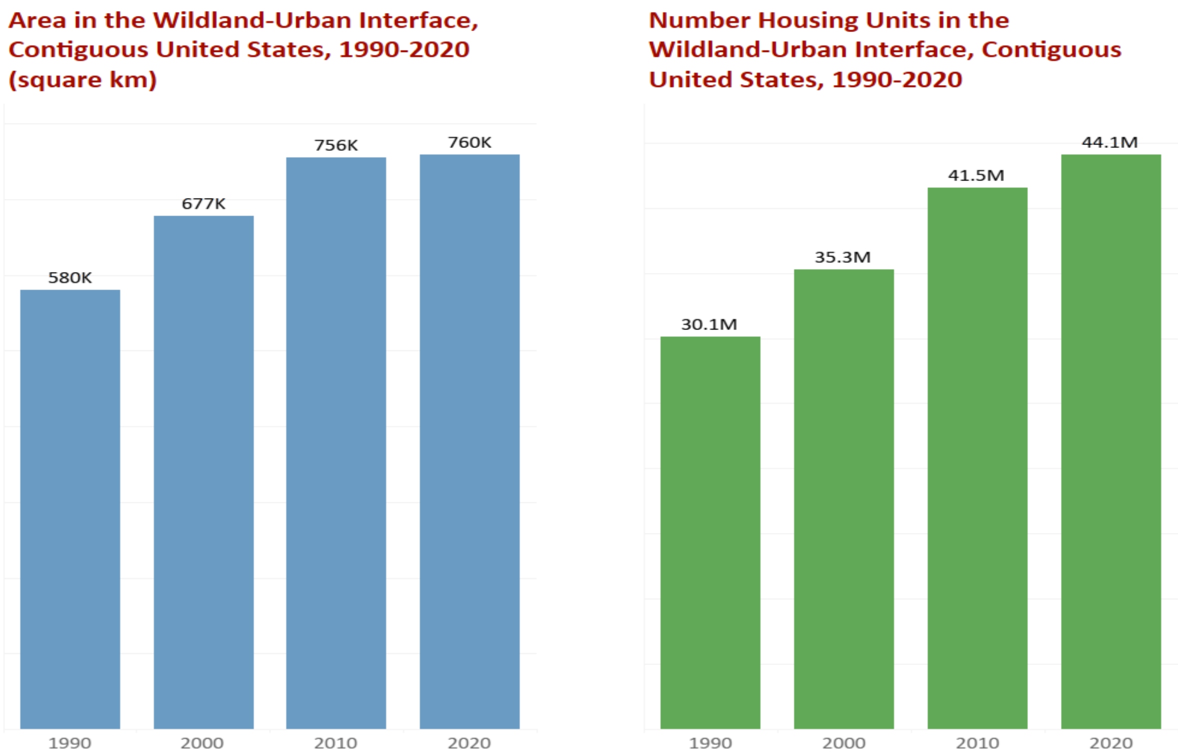
2.2. Managing invasive plants within the Wildland-Urban Interface

In the United States, urban areas are dominated by concrete, asphalt, and other generally barren surfaces. Suburban areas are developed, sometimes rapidly, extending the impacts of human activities well beyond the concentrated core of urban city centers. Some plants manage to live within these environments, often non-natives planted intentionally for agriculture, horticulture, aesthetic reasons, or ease of maintenance. And of these, some spread rapidly and become "invasive" to an area – such as English Holly, planted in the Pacific Northwest for aesthetic and commercial purposes (Stokes et al., 2014). In general, invasive plants are aided by anthropogenic disturbance by gaining a foothold in areas where the native plants are not able to recover as quickly (Simberloff et al., 2012).

This relationship between invasives and human development is most apparent beyond the urban zones, out in the wildland-urban interface (WUI) where housing intermingles with wildlands (Mockrin et al., 2023). Much of the lands within the WUI could be considered ex-urban or rural. Between 1990 and 2020, the WUI in the U.S. expanded by 31%, an area the size of Washington state, and housing within the WUI increased by 47% during this period (Figure 1). In 2020, the WUI contained 32% of housing in the contiguous U.S., amounting to more than 44 million homes (Mockrin et al., 2023).

Figure 1

Housing in the Wildland-Urban Interface from 1990-2020

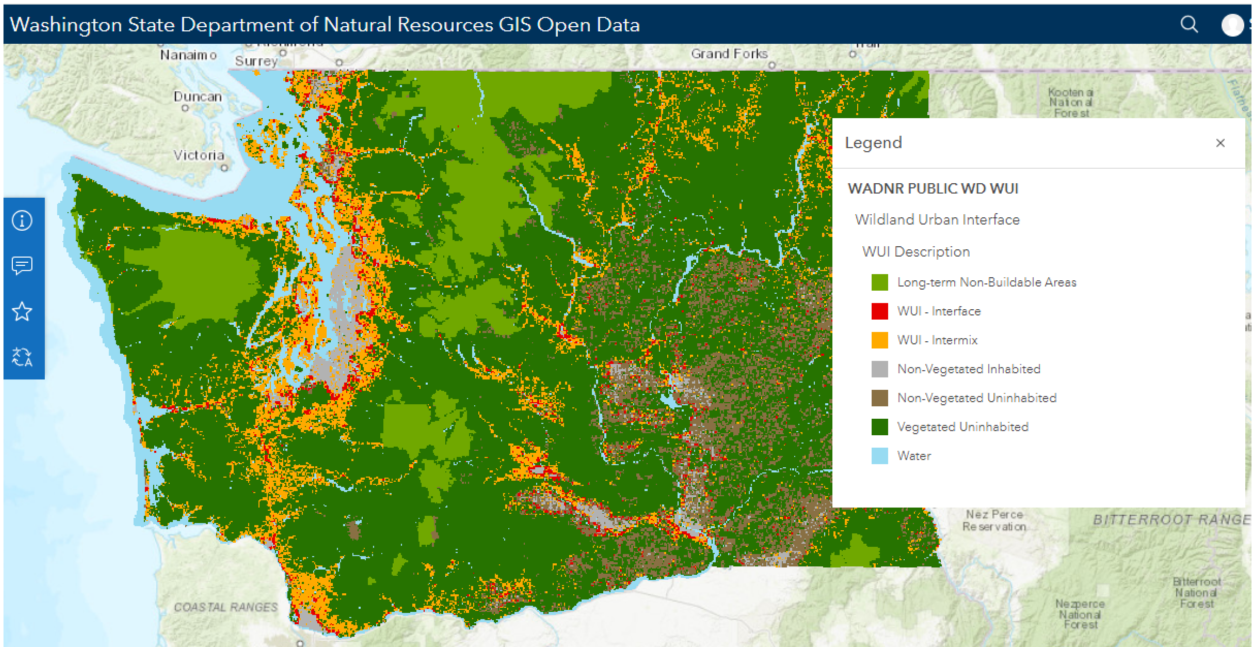


Note. The left chart shows the increase in area within the WUI, and the chart on the right shows the increase in housing units constructed in the WUI (Mockrin et al., 2023).

Expansion of the WUI is facilitated by sprawling suburban development, which heavily degrades natural habitats and compresses the boundaries of rural and wildlands, and ultimately enables establishment of invasive plants in a wider distribution of disturbed areas. A map of Washington's WUI shows much of these areas are along the I-5 corridor, a major interstate connecting most of the largest cities in the state (see Figure 2).

Figure 2

Map of WUI in Washington State

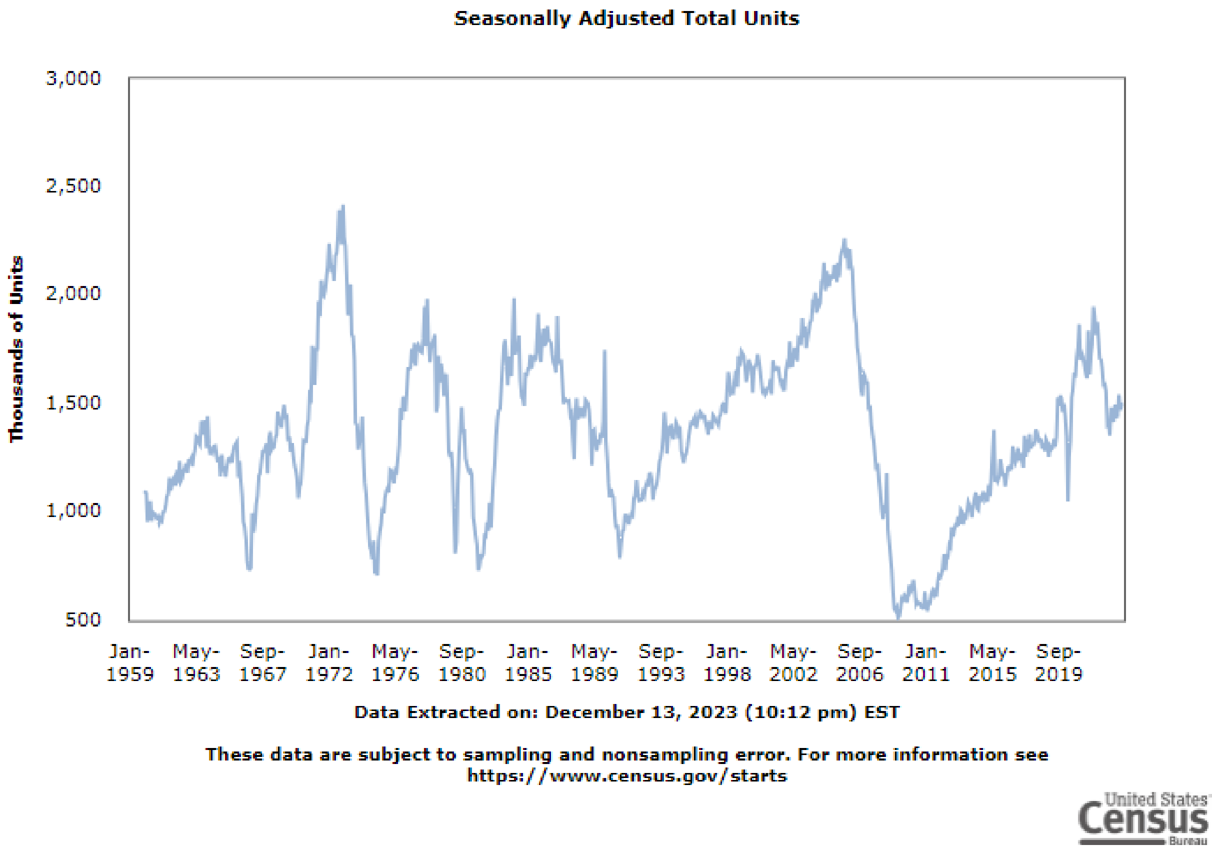


Note. Two types of WUI, interface and intermix, are highlighted in red and orange (Washington Department of Natural Resources, 2024).

Since the post-World War II housing boom in the 1950s, the rate of new housing construction has had many peaks and valleys, see Figure 3 (U.S. Census Bureau, n.d.). But around the early 90s, the rate of new homes being permitted steadily increased for 20 years, until the 2008 Recession. The market recovered a few years later, and housing construction is currently trending in a similar fashion, steadily increasing. Perhaps the most staggering piece of data in Figure 3 is that new housing construction has not fallen below 500,000 new permits a year, even during the Recession. This economic development pressure on the WUI will likely continue without aggressive planning and policy. An example of how policy could be leveraged is proposed by Kremen and Merenlender (2018), who advocate for the consideration of "working lands" for their conservation value.

Figure 3

Annual Housing Units Authorized in the United States



Note. Line graph of New Residential Construction data from the US Census Bureau from 1959 to 2023 (U.S. Census Bureau, n.d.).

The WUI includes, or is adjacent to, what can be considered "working lands" – used for agriculture, timber, and raising cattle (Kremen & Merenlender, 2018). Compared to the degradation caused by housing development, working lands provide some level of quality habitat for wildlife, and are becoming more recognized as vital to the conservation of biodiversity across the world (Kremen & Merenlender, 2018). However, working lands within the WUI are vulnerable to conversion to development (Kremen & Merenlender, 2018). Invasive plants can compound this pressure, adding to the inherently challenging tasks of farming, forestry, or maintaining rangelands. Managing invasive species, whether they be on working lands or pockets of preserved or public lands, is an iterative process, and heavily dependent on priorities and resources available. Restoration projects can attract funding for this type of work to landowners or organizations working on managing invasives, among other goals, and help them mobilize other resources such as volunteers.

Managing invasive plant species confronts us with philosophical and practical issues. Philosophically, it is difficult to measure impacts because like all conservation, invasion ecology is value-laden and based on the priorities of the people engaging in its practices (Baumgaertner & Holthuijzen, 2016). Different worldviews yield different perspectives on how to interact with other species, plants included (Sax et al., 2023). For insight into an Indigenous perspective on invasives management in Australia, see Bach et al (2019). Western anthropocentrism believes Nature must be managed to suit the needs of humanity. Even when the goal is conservation, people attempt to choose which species are conserved and which are controlled or completely eradicated, though this is sometimes an exercise in futility.

Practically, the most effective way of managing species deemed as “invasives” is prevention, which is costly and difficult to implement. The spread of invasive plants to a degree where an ecosystem experiences negative effects can, in theory, be prevented (Simberloff et al, 2013). Preventing invasives from establishing themselves in a new environment is the most effective management strategy; the next best is early detection and rapid response because eradication is not always viable (Simberloff et al, 2013). In places where invasion has occurred, and habitat degradation is evident, suppression of invasives and replanting of native species can reintroduce higher biodiversity and restore ecosystem services provided by those native plants (Gaertner et al., 2012). However, if climate adaptation is a goal, there may be value in better understanding the ecosystem services provided by invasive species, which may be more resilient to climate change than native plants (Sax et al, 2023, Sorte et al., 2013, and Howard, 2019). Given these potentialities, answering the question of whether investment in invasives management should continue on a broad scale is difficult.

2.2.1. Challenges for invasive plants management

Invasive plants management spans every level of society. International cooperation is required to reduce the transport of non-native and potentially invasive species (Reaser et al., 2020), reinforcing sociopolitical borders, but an individual can decide what to plant in their yard, so long as they are able to order seeds or propagules over the Internet. From a systems thinking perspective, invasive plants management falls into the tragedy of the commons trap, where the feedback (degraded habitat) is too delayed to elicit the desired outcome (proactive management) required to maintain shared resources (Meadows, 2008). An argument could be made that the feedback is actually displaced, rather than delayed, since some communities are insulated from ecological shifts to some degree. Well-resourced communities have the means to import species yet remain somewhat buffered against negative impacts. Communities dependent on biological resources for their livelihood are significantly affected by ecological changes (Howard, 2019), and, in many cases, left to deal with the consequences with little to no support.

Within the U.S., funding and labor are two extremely limiting factors in restoration work. These types of jobs are typically demanding, seasonal, entry level and often require working with toxic substances, making it difficult to attract and maintain a workforce. According to the U.S. Bureau of Labor Statistics, Forest and Conservation Technicians made an annual mean wage of \$36,700, and Environmental Science and Protection Technicians had an annual mean wage of \$43,150 (2012). Wages have hopefully increased since this data was collected in 2011, but the Green Goods and Services Occupations program was discontinued in 2013 due to budget constraints. When it comes to funding for this type of work, what is spent on management is a small fraction of the costs associated with invasive species. According to the Washington Invasive Species Council, a program of the Washington Recreation and Conservation office (RCO), a handful of invasive species, which includes HBB, have the potential to cause an annual loss of $1.3 billion in revenue and jobs related to recreation, water facilities, agriculture and forestry (2019). In contrast, the Recreation and Conservation Funding Board, managed by RCO, which helps finance these types of projects all over the state through grants, was allocated just over $17 million for 2024 (Office of Financial Management, 2024). This vast gap in funding leaves much of the financial burden to communities vulnerable to the impacts of invasive species.

2.2.2. Study species

2.2.2.1. Himalayan blackberry (Rubus armeniacus)

Despite the exotic name, Himalayan blackberry (*Rubus armeniacus*) is native to Western Europe and was brought to North America in the late 1800s for cultivation. HBB is a hardy perennial shrub in the Rose family (Rosaceae), with large, toothed leaves that are somewhat evergreen, and grouped in sets of three on side shoots or five on main stems. They are notable for having robust thorny stems. Shoots will grow as individual canes that can reach nine feet tall, and groups of canes can grow into huge mounds or banks. Canes that trail along the ground can spread 20-40 feet, rooting from the tips. In the second year of growth, white flowers turn into shiny, black, aggregate berries, ripening midsummer through autumn, later in the season than native blackberries (Soll, 2004). These berries tend to be bigger than native blackberries as well.

The Pacific Northwest is a fertile climate for HBB, particularly on the western side of the Cascades. HBB became naturalized on the West Coast by the 1940s and is able to grow nearly everywhere. In both alkaline and acidic soils, especially in areas with at least an average annual rainfall of 29 inches, and at altitudes up to 6000 feet. When left to thrive, it can form impenetrable thickets, cutting off corridors for both wildlife and human traffic. Other areas where HBB can spread rapidly include disturbed landscapes, pastures, forest monocultures, transportation infrastructure, riparian areas, and fence lines. While individual canes only live two to three years, they can densely sprout 525 canes per square meter, and produce a thicket five meters across (Soll, 2004).

There are multiple ways for HBB to spread. Root crowns can grow up to eight inches, with lateral roots and occasional adventitious shoots, which can sprout from 18 inches below the surface. HBB can propagate from both root fragments and cane cuttings. Thickets can produce over 10,000 seeds per square meter, and viable seeds occur nearly every year. However, dense shade will prohibit seed development in most blackberry species. Many species are known to disperse seeds, including omnivores like bears, coyotes, and humans, as well as birds. Dispersed seeds are viable for many years, though rarely germinate in the first spring after formation (Soll, 2004). Seed dispersion and propagation make complete removal of HBB difficult to achieve.

Biologically, HBB outcompetes other shrubs and once established, can repress seedling establishment of shade intolerant trees, such as Douglas fir. The ability for this plant to spread and suppress other plants poses a serious ecological threat. Dense HBB thickets, if left unchecked, can restrict movement of wildlife between habitats, and reduce the quality of forage in those habitats. While eaten by many native species, HBB alone is not preferable to a diversified diet of native plants (Soll, 2004). Minimizing movement effectively reduces available habitat, putting additional pressure on ecological systems already strained by impacts of human development.

2.2.2.1.2. Methods for HBB management

While control is difficult, it is possible, but generally requires multiple iterations of two phases. The first phase is removing vegetation above the ground, and the second phase is removing or killing root crowns and bigger side roots. There are a few options for removing above ground vegetation including mechanical, either by hand or machine, or controlled burns. When it comes to long-term control, there are six primary options (see Table 1). Yet another option is prescribed grazing, which could employ horses, cattle, goats or even chickens, and is effective to varying degrees (Soll, 2004). However, depending on the habitat, introducing animals for grazing may not be a viable option.

Table 1

Long-term Control Methods for Himalayan Blackberry

Control method	**Approximate cost**	**Timing considerations**
Grubbing roots	Most expensive	Can be applied any time of year
Repeated cutting of vegetation	Expensive	Requires multiple years of treatment
Foliar treatment of resprouted canes	Relatively inexpensive	Most effective when applied in the fall after summer clearing
Treating freshly cut stumps with herbicide (spot treatment)	Relatively inexpensive	Same as above
Broadcast application of herbicides	Potentially least expensive	Late summer or fall
Dense planting of shade producing vegetation	Relatively inexpensive, but depends on habitat type	Takes the longest time to take effect, but potentially the most effective

Note. Adapted from *Controlling Himalayan Blackberry in the Pacific Northwest* (Soll, 2004).

Despite being the least expensive options, applying chemical control methods requires careful selection, application, and knowledge of herbicide regulations. Detailed information can be found in the USDA Herbicide Handbook, which is updated annually (Soll, 2004).

2.2.2.2. Reed canary grass (Phalaris arundinacea)

The second species being evaluated in this study is reed canary grass (*Phalaris arundinacea*). Unlike Himalayan blackberry, there is evidence of a native North American cultivar (Tu, 2004). However, the cultivar that has spread into wetlands and can create monotypic stands is most likely a hybrid of a cultivar from Europe (Maurer et al., 2003). To this day, RCG is cultivated throughout Europe and North America for livestock forage (Waggy, 2010). It is a fast-growing species, with high tolerance of variations in hydroperiods, and outcompetes native grasses who are less tolerant of climatic fluctuation (Maurer et al., 2003). A perennial, rhizomatous plant, RCG can form thick mats, excluding other plants, while growing stems up to 2 meters tall and flat blades reaching 2 cm wide and 0.5 m long (Tu, 2004). In the Pacific Northwest, RCG is in full bloom from May to June, and its panicles, or inflorescences, change from light green to dark purple, fading to a golden straw color when fruit is dispersed, see Figure 4 (Tu, 2004). There is a high likelihood RCG was intentionally planted at the study site as forage for dairy cows. Hydrologic characteristics of the site seem to have influenced the spread of this species into areas far beyond historical plantings, as much of the marsh and streams are banked with mats of RCG.

Figure 4

Reed canary grass at various stages of inflorescence

*Note.*Various stages of reed canary grass inflorescence, with early stages on the left and post dispersal on the right (Tu, 2004).

RCG is highly resilient and spreads robustly in response to disturbance. Tussocks support biodiversity, creating diverse microtopography for several species to grow in close proximity, but sediment from runoff, flooding, and erosion fill these gaps which creates ideal habitat for RCG (Maurer et al, 2003). Each inflorescence can produce approximately 600 seeds, which can be dispersed on animals, vehicles, and water. However within dense patches, successful establishment from seeds is low, and most persistent populations of RCG are likely spreading from rhizomes. For both seeds and rhizomes, the most common dispersal vector is water, though seedlings are unlikely to survive prolonged drought or flooding. Established stands are able to survive these conditions, and can rapidly outcompete native plant species, diminishing food stock for native wildlife (Tu, 2004).

2.2.2.2.1. Methods for RCG management

There are several best practices when it comes to managing reed canary grass. The two main factors influencing which practices to apply in the near term include whether herbicide can be used in a particular location, and the distribution of RCG. Other factors influencing management include overall management goals, resources available, and presence of healthy native vegetation (Tu, 2004). Persistent shade is effective at reducing spread, so long-term ecosystem strategies for managing RCG include establishing forests through pole plantings, planting circles which recreate edge effects, de-leveling to increase micro-topographic diversity, and placing coarse woody debries to support carbon cycling, soil flora, and plant succession (Antieau, 2001). The table below summarizes best practice recommendations for near-term management of RCG, but it is important to note that the presence of native vegetation will likely influence which treatment to apply. Reseeding and replanting post treatment will likely be required on sites with large patches or monocultures of RCG (Tu, 2004).

Table 2

Recommended Best Practices for Managing Reed Canarygrass in the PNW

Method	**Small, scattered patches**	**Distinct patches** (large enough to mow)	**Large patches** (up to several acres)	**Large monocultures** (hundreds of acres)
Dig out	✓	✓	X	X
Spot-spray / wick with herbicide	✓	✓	X	X
Spot flame	✓	X	X	X
Spot burn	X	X	✓	X
Cover with shade cloth, solarization, & mulching	X	✓	✓	X
Mow (+cover)	X	✓	✓	X
Mow (+herbicide)	X	✓	✓	✓
Herbicide	X	X	✓	X
Prescribed burn (+herbicide)	X	X	X	✓
Tillage and flood	X	X	X	✓

Note. Adapted from *Reed Canarygrass Control & Management in the Pacific Northwest* (Tu, 2004).

2.2.2.3. Recommendations on managing both species

The best management practices for both RCG and HBB have many similarities. In the near term, multiple years of mowing reduces the seed bank. Both HBB and RCG should be mowed or cut in the early summer (May or June), prior to flowering and seed dispersal (Soll, 2004, Tu, 2004). If mowing is the only control method used, RCG requires multiple mowings a year (5+), for 5-10 years, and should precede herbicide application, tillage, or the use of shade cloths (Tu, 2004). This is similar for HBB, multiple cuttings per year for multiple years and also precedes other treatments (Soll, 2004). For both RCG and HBB, applying herbicide is most effective in the summer or fall, when foliage is present (Soll, 2004, Tu, 2004). HBB produces slash, which can be left in habitat piles for animals, removed, or burned, but care must be taken to not spread propagules (Soll, 2004). Control is possible, but it will be a multi-step, multi-year process.

2.3. Mapping

The first section of this literature review delved into several ecological aspects of invasive species management. This section will highlight some of the ways technology can be leveraged for this use case by discussing the role of mapping in habitat restoration and technology required. Developing a comprehensive picture of land cover and vegetation types throughout a landscape is an instrumental management tool for planning of all kinds. With the ability to create high spatial resolution maps, land managers can see the various landscape features which affect their management goals. Remote sensing enables planning across broader areas, across a greater variety of landscapes than ground observations. Using remotely sensed data, organizations across the world can begin to develop data-rich geographic information systems (GIS) to display land use and cover, and with regular updates from both remote and on the ground datasources, track changes over time.

Remote sensing of individual plants or differentiating between species is still in the experimental phase. Two avenues for interpreting remotely sensed imagery exist, manual and machine learning. Both methods require software to process imagery into orthomosaics. Orthomosaics are made up of orthoimages, which have geometric integrity and can be combined into a mosaic of map-quality imagery (What is photogrammetry?, n.d.). A commonly known example of an orthomosaic is Google Earth. Manual interpretation requires basic GIS skills such as drawing polygons, navigating while drawing, and controlling zoom extent. Machine learning requires coding knowledge and field samples to train computer models. However, it is time-consuming and tedious to walk hundreds of acres to assess the spread of a certain species, and in some landscapes, dangerous or impassable. If imagery could be analyzed for this data instead, mapping a restoration project can save time and money. Measurement at the landscape scale can be used for budgeting and work planning. Building a GIS of this kind would enable land managers to monitor management practices over time, gauging effectiveness based on the reduction of coverage of the managed species.

This process of differentiating and labeling objects in remotely sensed data is referred to as classification. Automating the classification of land cover types with machine learning has unlocked new levels of geospatial analysis, and providing open access to tested code is helping researchers get closer to making species-level classification a reality (Gatis et al, 2022). Many studies have applied machine learning models such as Random Forest to a variety of species across different landscapes to assess their accuracy (Dash et al, 2019). This study will not apply machine learning, but for further reading, see reviews by Neyns & Canters (2022), Dash et al. (2019), and Bradley (2014). For this study, I have chosen to do manual visual interpretation of the available remotely sensed data, as it requires less technical expertise. This method can be conducted by land managers who are very familiar with a particular species, but only requires a basic understanding of mapping software.

2.3.1. Technologies

Advances in remote sensing technology have progressed rapidly in recent years. Technologies such as radar, sonar, and satellites have military origins, and were declassified in the 1960s after World War II (Moore, 1979). The U.S. National Aeronautics and Space Administration (NASA) and U.S. Geological Survey (USGS) have been taking satellite images of the Earth's surface since 1972, a dataset referred to as Landsat, which is publicly accessible (Moore, 1979). This is notable because images of the entire Earth's surface have been taken on a regular basis, every 16 days, for over fifty years (U.S. Geological Survey, n.d.). While this is an impressive and rich dataset, it is limited to moderate spatial resolution, ranging from 15 to 100 meters per pixel (NASA, n.d.). With the emergence of private satellites, there is a growing market of satellite imagery providers, but there is a correlative relationship between resolution and cost, where the highest resolutions, down to mere centimeters, are cost prohibitive for smaller plots (Sozzi et al., 2021). Wider access to higher-resolution imagery has become commercially available in drone imagery.

Drones offer an affordable alternative to manned aircraft. In comparison to manned aircraft, there are lower barriers to access and regulation. Generally, manned aircraft must depart from airports, as opposed to the versatility of deploying drones from virtually anywhere. In the United States, flying drones requires registration and coordination with the Federal Aviation Association (FAA, 2024), but the costs and resources required to fly drones are far lower than manned aircraft.

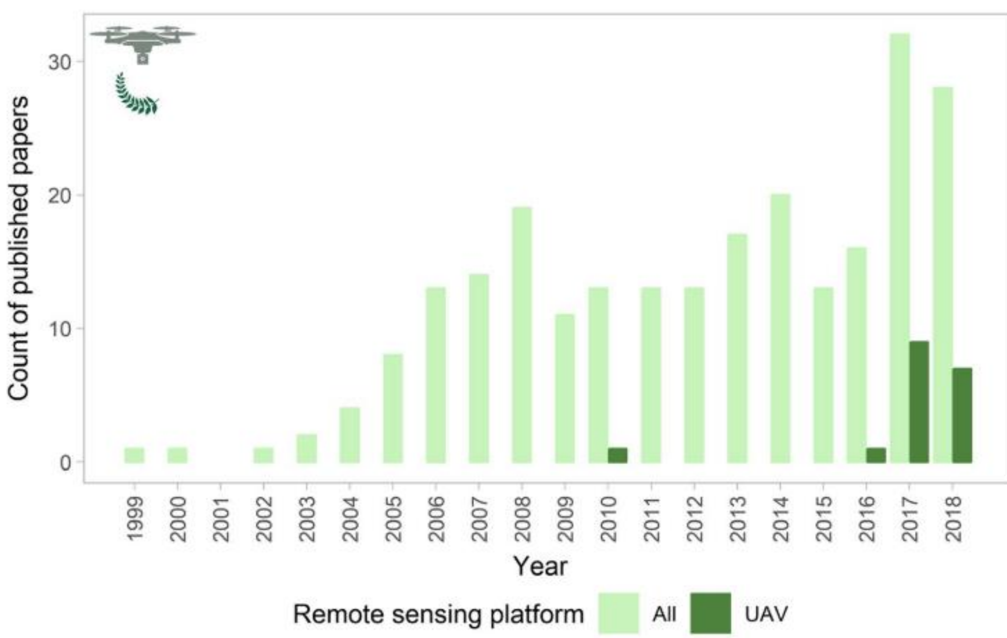
There are some limitations to using drones for this type of data collection. Depening on the location, there may be legal restrictions to protect commercial and urban air spaces. A drone's range is dependent on battery life, which may require subsequent flights, which can be further complicated by changes in lighting and weather (Dash et al., 2019). Technology can sometimes spontaneously fail, and the mission may need to be flown multiple times to collect all necessary imagery. But many of these limitations also apply to manned aircraft. With either platform, it is important for land managers to understand whether a company or contractor hired to collect imagery will also process this data into orthomosaics as the final product, or provide "raw data" in the form of hundreds or thousands of individual image files.

Drones are relatively novel technology, and research into the application of drone imagery for identifying invasive plants is less than a decade old. A literature review (Dash et al.,

2019) of invasive plant research found that out of 309 publications involving remote sensing technologies published between 1999 and 2018, less than 8% included drones (Figure 5). A relatively new area of study, all were published in the last 35 years, with the earliest drone study published in 2010. A surge in publications between 2016 and 2018 is linked to the commercial availability of durable, dependable drones, as well as increased awareness of the impacts of invasive plants. Initial applications of drones for invasives monitoring occurred in agricultural environments, as the accuracy of detection is best where characteristics of the surrounding vegetation vary significantly from the invasive plant being studied. Many other habitat types have since been studied, across several different countries and in both northern and southern hemispheres (Dash et al., 2019). Detection accuracy is greatly affected by the characteristics of the environment being studied, so it is crucial to continue this research in different locations.

Figure 5

Remote Sensing Publications in Invasive Plants Research



Note. As of 2018, only 8% of all remote sensing studies of invasive plant research used UAVs/drones (Dash et al., 2019).

There are two types of drones, based on wing configuration, and two groups of mounted sensors, active and passive. Fixed-wing craft can have longer, faster flights while rotary wing craft are more maneuverable, capable of hovering, allowing for inspection work and more complex flight configurations (Dash et al., 2019). Rotary wing craft are more commonly used, likely due to the availability of reliable and low-cost models, but they are also easier to control, reducing risk of losing or damaging the mounted sensor (Dash et al., 2019). The sensor is typically the most expensive part of the system. Active sensors emit electromagnetic radiation and measure reflectance, while passive sensors measure solar radiation reflected by surfaces being measured in the visible and near-infrared range (Gambardella et al., 2021). Within each of these groups, the resolution that can be achieved by different sensors varies greatly, ranging from high spatial resolution of 1-5 meters to very high spatial resolution of 100-7.5 cm (see Table 3) (Neyns & Canters, 2022). Spectral resolution refers to detecting across bands of light, and some sensors can generate multispectral, hyperspectral, and LiDAR data (Neyns & Canters, 2022).

Table 3

Sensors Used in Vegetation Classification Studies

Sensor	Spatial Resolution [m]	Spectral Resolution [# Bands]	Classification Scheme
		High spatial resolution [1–5 m]	
Gao-Fen 2	4	4	green space
IKONOS	4 (MS)	4	vegetation communities, tree species
Quickbird	2.62 (MS)	4	green space
Pleiades	2 (MS)	4	tree species
Rapid-Eye	5 (MS)	5	green space, plots of homogeneous trees
Worldview-2	2 (MS)	8	green infrastructure, tree species
Worldview-3	1.24 (MS) 3.7 (SWIR)	16	tree species
CASI	2	32 (429–954 nm)	vegetation types, tree species
AISA	2	(400–850 nm)	tree species
HyMap	3	125	tree species
Hypex VNIR 1600	2	160	green infrastructure
AISA	2	186 (400–850 nm)	tree species
APEX	2	218 (412–2431 nm)	functional vegetation types
AVIRIS	3–17	224	tree species
AISA+	2.2	248 (400–970 nm)	tree species
AISA Dual hyperspectral sensor	1.6	492	tree species
		Very high spatial resolution [≤1 m]	
Nearmap Aerial photos	0.6	3	tree species
Aerial photos (various)	0.075–0.4 (RGB)	3	vegetation types, tree species
NAIP	1	4	functional vegetation types
Aerial photos (various)	0.20–0.5 (VNIR)	4	tree species
Air sensing inc.	0.06 (VNIR)	4	tree species
Rikola	0.65	16 (500–900 nm)	tree species
Eagle	1	63 (400–970 nm)	tree species
CASI 1500	1	72 (363–1051 nm)	shrub species

Note. Bolded sensors produce an additional panchromatic band which achieves resolution below 1 meter, and italiscized sensors were airborne, as opposed to satellite (Neyns & Canter, 2022).

2.3.2. Mapping approaches

Most papers on the topic of mapping vegetation with remote sensing have been published in the last decade, highlighting the recent advances in technology and application among researchers. Vegetation typology is a significant factor in mapping, and generally, split into two general categories: functional or taxonomy. Functional vegetation type, also known as plant functional type (PFT), is a commonly used remote sensing term which categorizes plants based on their ecosystem function and resource requirements, i.e. woody or herbaceous (Neyns &

Canters, 2022). Urban vegetation is more difficult to assess compared to wildlands due to higher fragmentation (Feng et al., 2015), and a lack of accurate classification methods, whether manually or machine learning, for species smaller than trees (Neyns & Canters, 2022). Classification methods are used to train computer programs which analyze remotely sensed imagery for estimated locations of the target species (Masocha & Skidmore, 2011). Researchers have had the most success with mapping large tree species and have been able to leverage satellite imagery with a spatial resolution anywhere between one and five meters, compared to what is considered very high spatial resolution, which would be less than one meter (Neyns & Canters, 2022).

2.3.3. Monitoring

Monitoring a species within a habitat restoration project serves a multitude of objectives. It is important to note that some species are impossible to track or measure their population, and for the ones that are possible, it is no easy task. Ecologists must be able to identify relationships between biotic and abiotic features and would ideally be able to evaluate them at multiple spatial scales (Mack et al., 2007). Remote sensing provides a great range of spatial scales, and when paired with a temporal scale, enables monitoring over time. Tracking visible species, or the effects of their presence, over time provides valuable information about population, spread, and extent for land managers (Mack et al., 2007). This type of monitoring can also provide useful information about work completed over time, e.g. replanting or removal of a species, and planning for future goals.

Getting a bird's eye view of an area enables land managers to make habitat-scale decisions, but historically, acquiring or generating this data was either not possible or cost-prohibitive. Monitoring can support restoration work in measuring impact of both an invasive species but also efficacy of management practices over time. This is valuable evidence which can be used as justification for investment or continued funding. Monitoring could precede funding, i.e. supporting evidence on the severity of need for restoration, but this would require having accesss to other resources to accomplish this work. But, having this sort of data can provide a quantifiable and possibly persuasive case for why funding should be directed to a project. In the case of cyclical funding, monitoring provides data on efficacy of past years efforts and can justify continued investment. For non-profits engaged in restoration, this can come from citizens of the community who donate to support their mission. Unfortunately, due to the expense associated with field surveys, allocation for monitoring is often rejected by funding agencies (Lishawa et al., 2017). Remote sensing may offer less expensive avenues for monitoring in the future.

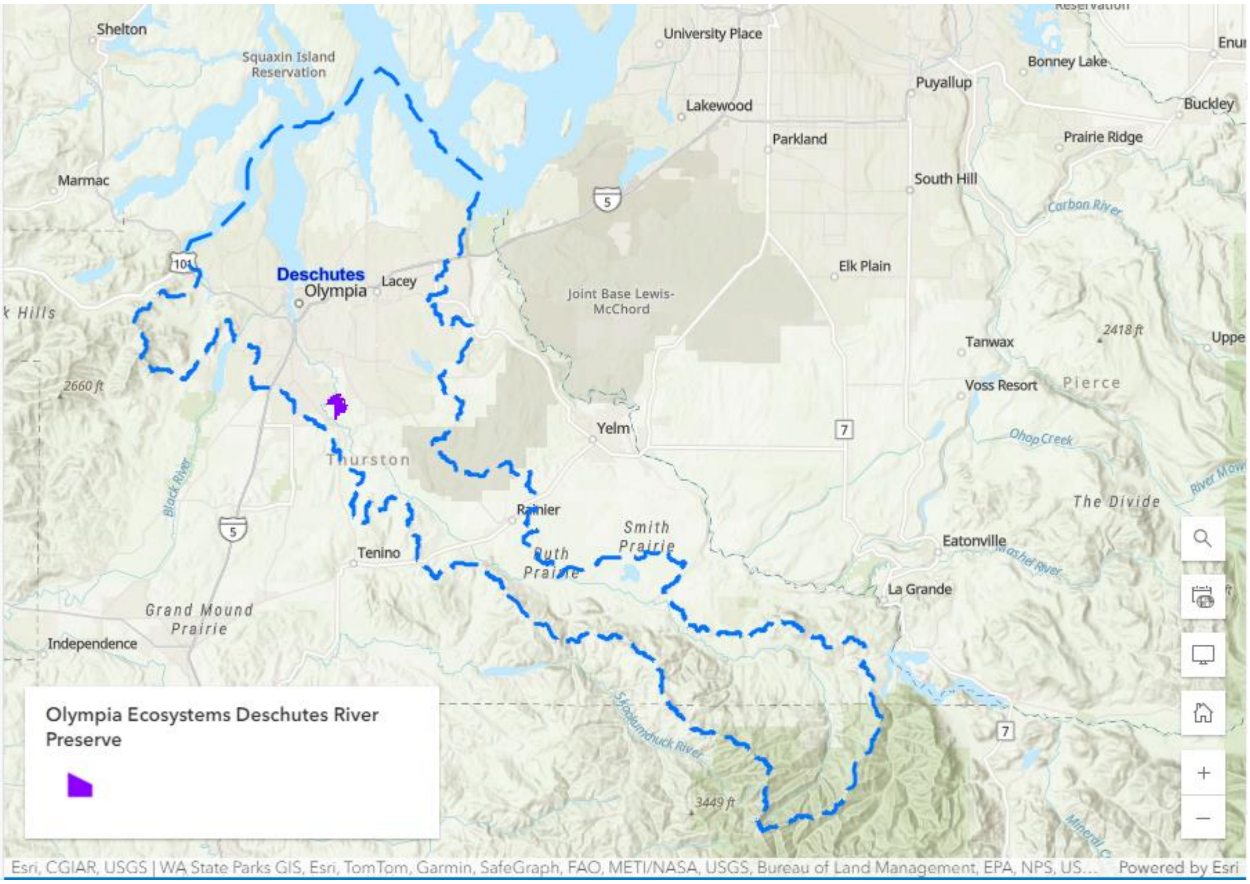
2.4. Study Site

As stated in the introduction of this literature review, the study site for this research is the Deschutes River Preserve (DRP). DRP is in southeast Olympia, Washington, and was purchased by OlyEcosystems in 2022 (OlyEcosystems, 2023). The intention of the organization is restoration and conservation of natural habitat in perpetuity. It is within the Deschutes Watershed, which is approximately 26% urban or designated urban growth area (OlyEcosystems, 2023). The Deschutes River and its tributary creeks do not meet clean water standards and are considered “impaired waters” by the Environmental Protection Agency (Washington State Department of Ecology, n.d., United States Environmental Protection Agency, 2021). The extent of development, and resulting pollution, has reduced wildlife habitat, diminished water quality, and catalyzed aggressive erosion along riverbanks (OlyEcosystems, 2023). These negative impacts, as well as the deleterious effects of stormwater runoff on aquatic species, contributed to the dramatic decreases in the Coho salmon population in the Deschutes Watershed (OlyEcosystems, 2023). Offsetting the impacts of human development, namely ecosystem degradation (Millennium Ecosystem Assessment, 2005), through habitat restoration is critical for the conservation of biodiversity and building environmental resilience in the face of climate change.

Watershed planning in Washington state involves many different organizations. There are 62 Water Resource Inventory Areas in the state of Washington, under the authority of the Department of Ecology, who is primarily responsible for establishment and regulation of policies designed “for the beneficial use of public waters” (Title 173, 1976). WRIAs are committees comprised of citizens and considered the “lead entity” when it comes to managing salmon recovery in their watershed (Water Resources Program, 2022). The Deschutes watershed is WRIA 13 and the committee roster from 2022 included representatives from counties, municipalities, tribes, utility companies, business organizations, non-profits, and several state agencies (Water Resources Program, 2022). Because WRIA 13 manages public waters and OlyEcosystems owns property adjacent to the Deschutes River, there is alignment in goals for salmon recovery and habitat management.

Figure 6

Map of WRIA 13 Deschutes Watershed



Note. WRIA 13 is outlined in blue, and the Deschutes River Preserve is highlighted purple (Ruth, 2023 and Ruth, 2020).

While the Deschutes River provides natural northern and western borders for the preserve, there are two clusters of a neighborhood development to the east, and agricultural lands to the south. In 2005, a habitat management plan was drafted in support of the Keeneland Park project, presently called Keanland Park. This is a Planned Rural Residential Development that subdivided a 315-acre parcel in Thurston County into one 269-acre Resource Use Parcel and 99 single-family residential lots. A habitat management plan was required due to the sensitive critical habitat within the Resource Use Parcel, and presence of great blue heron (*Ardea* *herodias*), osprey (*Pandion haliaetu*), and what were potentially bald eagle (*Haliaeetus leucocephalus)* nests, all of which were monitored species in Washington state at the time. The authors of the habitat management plan consulted with the Washington Department of Fish and Wildlife to address potential impacts, include mitigation measures, and meet special report requirements of Thurston County Code 17.15.735. The plan also notes existing agricultural uses such as hay production and grazing would continue on an 80-acre portion of the Resource Use Parcel, which is towards the south end of the preserve (McGinnis & Krippner, 2008). The Resource Use Parcel contains multiple habitat types.

According to the habitat management plan, habitat along the northern edge of the preserve is upland forest comprised of mostly red alder (*Alnus rubra*) and big leaf maple (*Acer macrophyllum*), with some Douglas fir (*Pseudotsuga menziesii*) and western red cedar (*Thuja plicata*) totaling approximately 44 acres. Upland pasture, agricultural fields, and developed areas, including roads, covered the largest area at 166 acres. Wetlands make up about a third, 105 acres, and surrounding marshlands cover around 78 acres. Ayer Creek and Elwanger Creek spring from these wetlands and flow northward 900 feet to meet up with the Deschutes River. Spring, summer and fall Chinook salmon were recorded in Ayer Creek at the time of writing the habitat management plan. The habitat management plan mentions several other animal species found on the property, which is congruent with its agricultural history (McGinnis & Krippner, 2008). In many parts of the world, especially "developed" countries such as the United States, wildlife has had to adapt to finding suitable habitat within agricultural lands (Carlson, 1985). The next section will provide a brief description of the historical land use as well as how the preserve was conserved.

2.4.1. Historical land use and current status

It is difficult to locate records about the previous inhabitants, but the HMP states that until 2000, the land was used for dairy farming (McGinnis & Krippner, 2008). Western agricultural practices of the last century sought to create manageable monocultures of various “crop” species, including feed for domesticated animals raised as “livestock” which meant robust grass species, like reed canary grass (RCG), were brought to new areas (Waggy, 2010). It was noted in the HMP that RCG was the dominant plant species along the edges of wetlands (McGinnis & Krippner, 2008). The historical use of the parcel will not reveal exactly when RCG made its way to the palustrine areas of this land, but it was potentially introduced as grazing for cattle.

In 2012, Capital Land Trust applied to acquire the property with the goal of conserving critical riparian habitat from becoming more neighborhoods, and “create the largest contiguous, protected habitat area in the lower Deschutes Watershed” (Washington State Recreation and Conservation Office, n.d.). Pre-acquisition, this property was identified as a “High Priority” action in the Salmon Habitat Protection and Restoration Plan for WRIA 13, and for the Squaxin Island Tribe South Sound Watershed Action Conservation Plan, it was the highest protection priority within the lower watershed (Washington State Recreation and Conservation Office, n.d.). At this time, the Chinook population in the Puget Sound is considered “threatened” under the Endangered Species Act, and an Evolutionarily Significant Unit, as a Pacific salmon population with important genetic variation for the “evolutionary legacy” of the species (Washington State Recreation and Conservation Office, n.d., Office of Protected Resources, 2022). Capital Land Trust’s grant application went dormant, but a decade later, OlyEcosystems was able to acquire a large part of the property in 2022 (OlyEcosystems, 2023).

2.4.2. Restoration goals

As stated previously, OlyEcosystems, in collaboration with Capital Land Trust and the Thurston Conservation District, successfully conserved the property in perpetuity. This acquisition prevents further development, which would have significantly impacted intact wildlife habitat in this watershed if not conserved. Presently, OlyEcosystems and their partners are actively restoring habitat by removing invasive plant species and planting local species, which will increase the biodiversity on site. This work sets out to undo over a century of degradation caused by the agriculture that took place on this land (OlyEcosystems, 2023). Riparian habitats are crucial for a wide range of fish and many other wildlife species, and salmon in particular have a symbiotic relationship with the trees and shrubs along streams. Through isotopic analysis, researchers discovered nearly 25% of foliar nitrogen was derived from spawning salmon (Helfield & Naiman, 2001). This preserve is 65% riparian, with 150 acres of emergent and open water wetlands, providing important cold-water habitat for juvenile salmonids, which supports recovery of Coho in this part of the river (OlyEcosystems, 2023). Another possibility for this preserve, if managed appropriately, is aquifer recharge and flood mitigation for the cities of Olympia and Tumwater (OlyEcosystems, 2023).

OlyEcosystems, Thurston Conservation District, and Wild Fish Conservancy are collaborating on a multi-phase project to restore water quality and habitat using strategies such as beaver dam analogs and pile assisted log jams to increase water storage onsite, attenuate stormflow, and increase the complexity of flow paths and dissolved oxygen (DO) in the water. Phase I of this plan includes invasive weed control and planting in riparian areas. The overall goal of the plan is to address the water quality issues from nonpoint source pollution and monitor the effectiveness of treatments from 2025 to 2027. Tracking parameters such as temperature, pH, and DO, since these are used by the Department of Ecology for Total Maximum Daily Load

(TMDL) reports (D. Einstein, personal communication, November 27, 2023). Ultimately, this project will improve riparian and wetland habitats on the preserve, but more importantly, within a heavily developed watershed at the base of Puget Sound.

2.5. Conclusion

Extensive research has been done on control and management of HBB and RCG in the Pacific Northwest. Remote sensing and computing technology has been leveraged to support vegetation monitoring, especially when differentiating between woody and herbaceous vegetation from other ground cover. Studies have successfully identified specific tree species and patches of monocultures of a few plant species. Research is still needed to explore how well these technologies can be applied to detect as many different kinds of invasive plant species as is feasible, and across various habitats. A common theme in the literature is that despite current accuracy and success rates, field data is still required for assessing automated image analysis, so relying solely on remote monitoring is not yet recommended.

This study will analyze drone imagery taken during the growing season to ascertain an initial estimate of the abundance and spread of Himalayan blackberry and reed canary grass. Field surveys will be conducted to determine the accuracy of this initial estimate from drone imagery. Unfortunately, due to time restraints, a confounding variable will be introduced as the field surveys will take place during the winter when the plants are dormant. I predict that the difference in seasons may affect detectability during field surveys, but if the drone imagery assesses more coverage than field survey, this will provide insight into the viability of relying on drone imagery for detection. However, if detection is marginal or low for these species, land managers working to manage HBB and/or RCG likely need to rely on data collected on the ground until technology improves.

3. Methods

This study aims to determine the feasibility of using remotely sensed imagery for measuring the presence of two invasive species across various habitat types. Two different mapping approaches were applied to determine the distribution of Himalayan blackberry (*Rubus armeniacus*) and reed canary grass (*Phalaris arundinacea*) at the Deschutes River Preserve. First by using drone imagery to visually identify patches of these species via manual interpretation, I then collected this data on the ground. This allowed me to compare the total area of coverage, as well as the locations of specific patches, and computed how well the drone measurements overlay field measurements. This provides an opportunity to test the use of drone imagery in estimating the extent of these species across different habitat types. It also enables geospatial analysis such as calculating the area of individual patches, as well as the combined total area of all patches per species. This information is useful for restoration project managers when it comes to estimating resources required for addressing invasive plant species and can provide helpful data when applying for funding to accomplish this work.

In this chapter, I will outline how the drone imagery was acquired and other sources of GIS data used in this study. Next, I will describe my workflow, how I determined the most efficient zoom extent for identifying and drawing polygons around patches of the study species. Then, I spent several days conducting field data collection, walking the preserve and dropping points on ESRI's Field Maps app to create a new layer of polygons to measure the size and spread of patches. Finally, I will outline which geoprocessing tools I applied using ArcGIS Pro, comparing the layers to determine where polygons overlap. I also quantify what percentage of patches were either over or underestimated through imagery analysis.

The Deschutes River Preserve is a recently protected area is located within the Deschutes River watershed, and the river borders the north end of the property. Himalayan blackberry and reed canary grass are two quickly-spreading plant species, and RCG in particular can affect hydrology in addition to wetland habitat by creating thick mats that are difficult for other species to survive in (Maurer et al., 2003). After analyzing drone imagery to identify patches of these species, I surveyed the photographed areas of the study site to “ground truth” the estimations. By comparing the resulting layers of polygons which represent patches of HBB or RCG, I provide a case study for land managers who can decide if drone imagery is sufficiently accurate for the purpose of identifying these two species in their own projects. I also gained a sense for how much time it takes to analyze available imagery and collect this data in the field, which is another significant factor when it comes to resource allocation for restoration management projects. However, my research does not factor in the time or cost of acquiring drone imagery. Satellite imagery may also be sufficient, as well as freely available, and I will expand on this in the Discussion.

3.2. Drone imagery orthomosaics acquisition

For a land manager or GIS professional to analyze drone imagery, it must first be captured. Evergreen professor Mike Ruth captured drone imagery of the study in the spring and summer of 2023. He had already processed the images into several orthomosaic map layers, one for each date the images were captured, and they are available on ArcGIS Online (AGOL). Creating the orthomosaic is an important step between capturing drone imagery and having a useful GIS layer to analyze. This is ideally the type of product land managers would contract for when hiring a drone operator to take imagery of their property, as opposed to receiving unprocessed image files.

Habitat types at the study site have been mapped by Melinda Wood and accessed through AGOL (Wood, 2023). The habitat types used were based on visuals from the drone imagery, previous experience walking the site, and landcover data generated by Washington Department of Fish and Wildlife, who uses high resolution imagery collected as part of the National Agriculture Imagery Program (NAIP). The landcover layer has a 1-meter resolution and displays various types of vegetation, differentiating between trees and shrubs, water, and gravel (M. Wood, personal communication, December 12, 2023). According to WDFW, the data limitations of the land cover data make it less reliable for vegetation below 5 feet tall, which are classified as "herbaceous" land cover. Higher rates of error can also be expected when looking at "built" areas referring to buildings and roads, and "ground" areas covering rocks and gravel, all of which are likely to contain mixed classes in a single polygon. For example, a road could run through a field, but that polygon would be considered "herbaceous" and the area covered by the road would be obscured by this classification. This land cover data is most reliable for classification of "trees" or vegetation over 15 feet, "shrub" vegetation between 5-15 feet tall, as well as "water" and "gravel" which are derived from their Visible Surface Water data (Washington Department of Fish and Wildlife, n.d.). By collecting habitat type data for polygons, especially along transitions between habitats, the habitat types layer can also be tested against data collected in the field.

3.3. Manual interpretation of the drone imagery

My first step was to find the orthomosaic layers and habitat types layer in AGOL, and add them to a new map. In order to draw polygons, I created a new feature layer and labeled it "Drone Polygons" to simplify data management. With the habitat types layer toggled off, I zoomed in close enough on the orthomosaic to visually identify Himalayan blackberry plants, as well as reed canary grass, and drew polygons around them to mark their size and locations. There is limited functionality on AGOL compared to ArcGIS Pro, so to open the orthomosaic layers in the Pro software, I had to get copies of the TIF files from Mike Ruth on an external drive.

Manual visual inspection is somewhat subjective, and a trade-off between accuracy and time must be made when deciding where to set the map scale. Map scale is displayed as a ratio, where 1 unit of the map is equal to a specified area, i.e. 1:500 could mean 1 unit on the map is 500 meters on the ground. The drone imagery resolution allowed me to zoom into 1:50 meters but being that zoomed in made it difficult to draw polygons. Being zoomed in too close required more scrolling than when zoomed out to 1:2000 meters. At this zoom extent, entire patches of HBB were visible and drawing polygons around them required very little scrolling. However, at that level of zoom, it was difficult to differentiate between HBB patches and ivy, or other bright leafy green colors. In some areas of dense shrubs and trees, it was nearly impossible to differentiate at any zoom extent.

When scanning the drone imagery, I primarily looked for the red canes and large leaves of HBB, and fully senesced RCG patches are an identifiable shade of light gold. RCG patches were much larger and still required scrolling while drawing at the 1:2000 zoom extent. Another aspect of this workflow I wanted to quantify was the time spent on imagery analysis. This could be used as a point of comparison to conduct field surveys, and it took me 7 hours to review approximately 100 acres of the preserve, see Figure 7. Factors that could reduce time required include extensive drawing experience in ArcGIS Pro or greater confidence with species identification. Regardless of experience, the area of analysis would have a linear relationship with time needed for visual inspection, where greater areas will require more time.

Figure 7

Layer of Drone Polygons

Note. Polygons were drawn manually, through visual interpretation of orthomosaics of drone imagery. Yellow polygons represent RCG, and green polygons represent HBB.

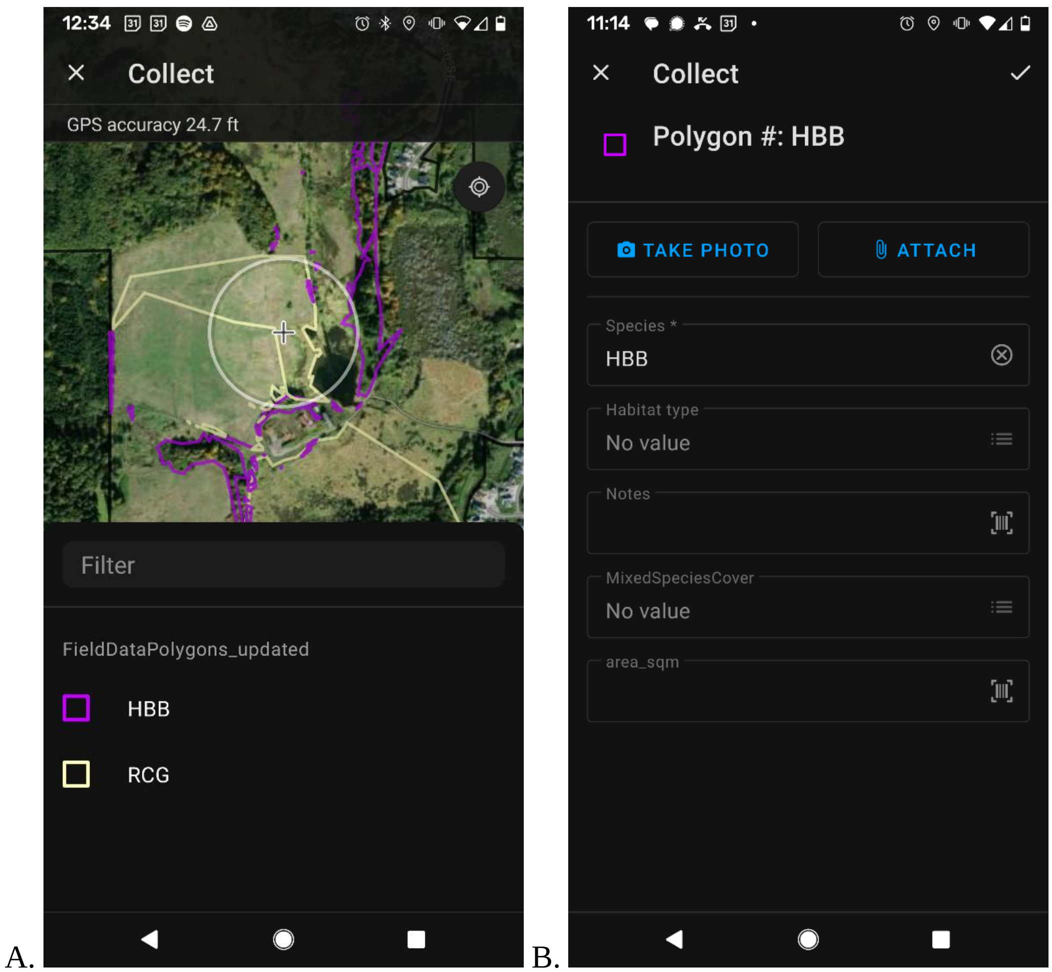
3.4. Field data collection and Field Maps schema

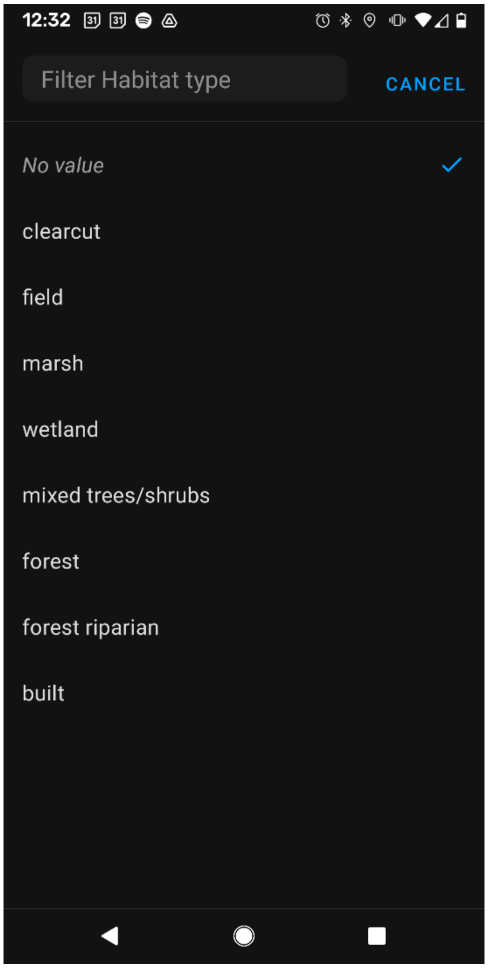
Before I could collect data in the field, I had to build an editable layer with additional fields besides the default fields for polygon features. Figure 8A is a screenshot of what ESRI's

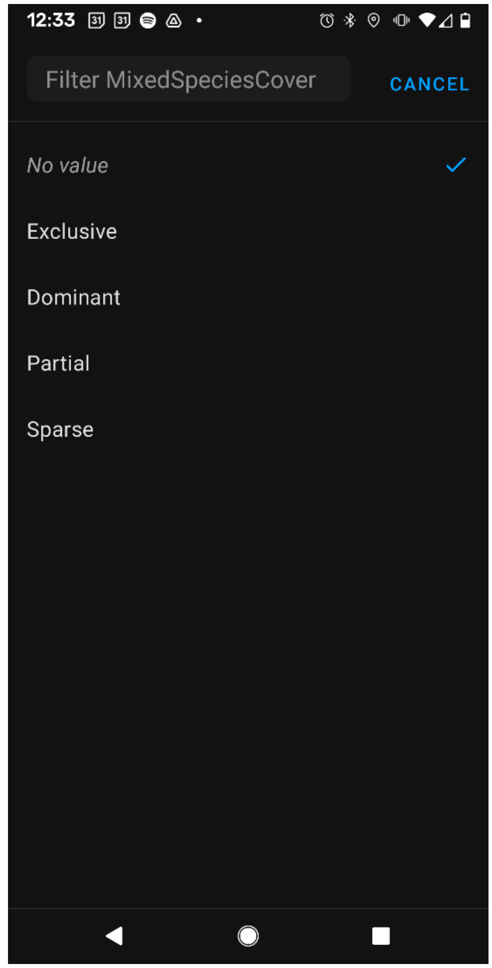
Field Maps application looks like, with the map I created to capture polygons open, and a point has been dropped on the map to begin collecting data. The enumerator selects which species this polygon will represent, HBB or RCG, and then the fields view appears to collect the additional data (Figure 8B). To better understand where HBB and RCG patches were located, I included a habitat type field, with pre-programmed options for enumerators. The options were based on the habitat types layer (Figure 8C).

Figure 8

Various Screenshots of ESRI's Field Maps Application



C. 

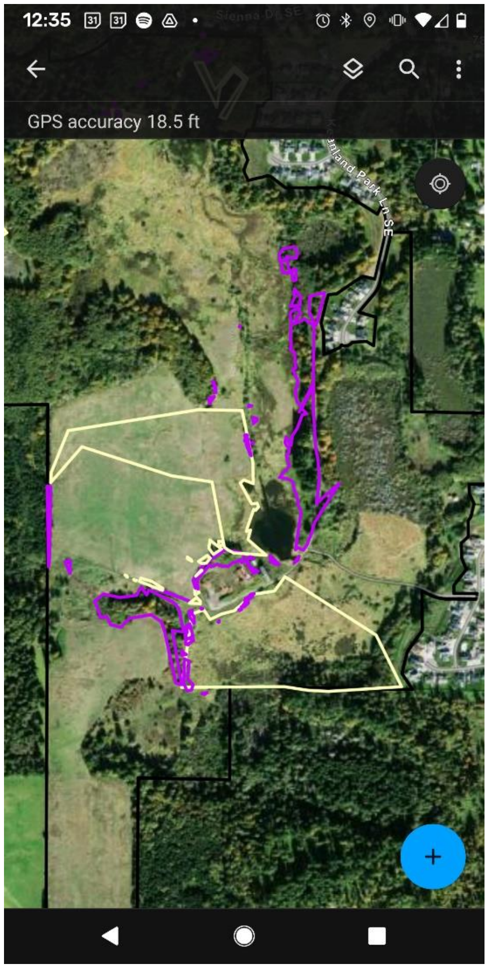
D. 

Note. A. Shows what options appear once a point is dropped, and data collection is initiated. The first question is a choice of species, HBB or RCG. B. Displays the fields that appear once a species selection is made. C. Lists the habitat type options. D. The "MixedSpeciesCover" field selections.

In the notes field, I commented on other species present or landmarks that influenced the shape of a polygon. Photos were taken of the patches at various points to accompany the "MixedSpeciesCover" field, which classified the amount of coverage of the invasive species, whether it was the only species present, dominant, partially covering the patch, or sparse (Figure 8D). Throughout surveying, the data is always visible, which makes collecting simultaneously possible without duplicating polygons (Figure 9).

Figure 9

Screenshot of Data Collection Map on Field Maps

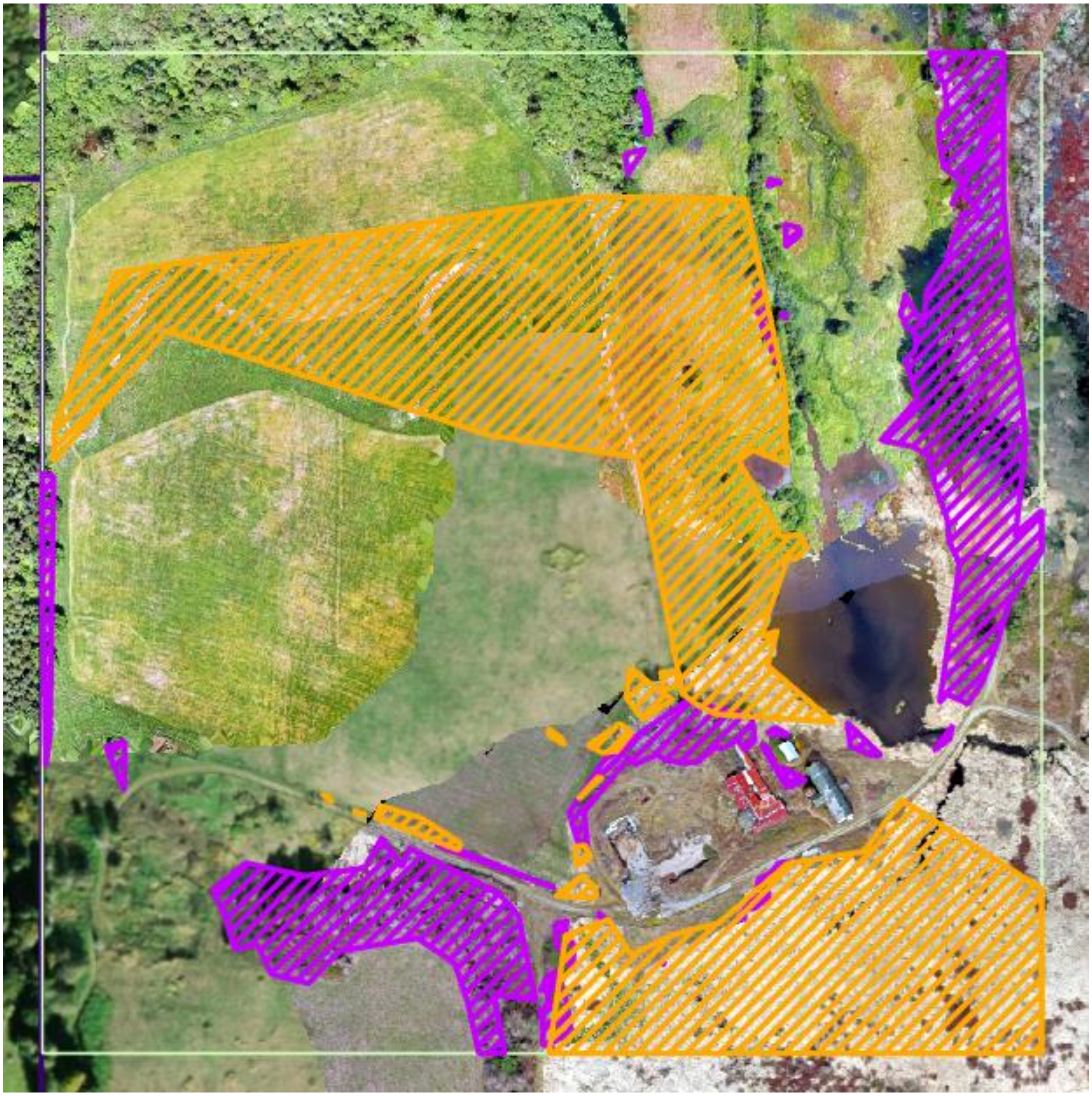


Note. Screenshot of map view with collected data of field polygons for each species, purple polygons represent HBB and yellow polygons represent RCG. Because data collected is displayed in real time, enumerators can see which polygons have been collected.

Throughout the months of February and March 2024, I visited the study site multiple times to walk the preserve, with the help of Courtney Murphy and Maddie Thompson, fellow MES students. While walking, we identified patches of HBB and RCG, and plotted their locations by using Field Maps (Figure 10). Enumerators dropped points while walking the perimeter of a patch, generally adding a new point where turns or slight changes in direction were made. We did not cover the entire preserve, as I narrowed the scope of this research based on the standing water and time available, but we covered approximately ### acres. This process of surveying the site took approximately 11 hours.

Figure 10

Layer of Field Polygons



Note. Field polygons based on data collected in the field using ESRI's Field Maps application. Orange polygons represent RCG and purple polygons represent HBB.

3.5. Comparing data collection methods

With both layers of polygons on the same map, I could visually see where the polygons overlapped. This provided a rough estimate of how much of the field collected data was visually identified in the drone imagery, but there are several tools within ArcGIS to do further analysis. Because these are polygon features, the area of each shape is calculated automatically as it is drawn, and by using the Calculate Geometry tool, I could convert the measurements into meters, for consistency and accuracy when running subsequent geoprocessing tools. The Summary Statistics geoprocessing tool calculated the total area of all the polygons. This provides the data necessary to calculate an overlap percentage.

To calculate the total area of overlap, I used the Pairwise Intersect tool, which produces a new feature class of polygons based on the areas of overlap between drone imagery polygons and field data polygons (Figure 11). Essentially, this tool highlights the overlaps between polygons and quantifies the area of overlap. With this data, the ratio of overlap can be calculated using total area of field polygons as the denominator, comparing the total area of intersection to total area of field polygons. This ratio can then be converted to a percentage, which is a more understandable representation of how much overlap exists between drone and field polygons.

Figure 11

Layer of Intersect Polygons

Note. The areas of overlap between drone and field polygons for RCG are represented as light yellow polygons, and the light purple polygons represent HBB.

Another way to analyze the drone polygons in relation to the field polygons is a percent change calculation. Taking the difference between the total area of drone polygons and field polygons, dividing by the total area of field polygons, then multiplying by 100 yields the percent change drone polygons have compared to the field polygons. This percent change calculation includes all drone polygons, not just the areas of intersection, reflecting a more inclusive percent difference compared to the percent overlap.

4. Results

Through manual interpretation of the drone imagery, I created a feature class layer of green and yellow polygons, representing Himalayan blackberry and reed canary grass respectively. I will refer to these as “drone polygons” through the rest of this paper. I also visited the study site to collect data in the field, generating a second layer of polygons (“field polygons”), using purple for HBB and orange for RCG. Due to inclement weather and difficulty traversing the preserve during the wet season, the scope of the study was reduced from the entire preserve to a subsection. Drone imagery was collected across multiple seasons in 2023, during February, March, May, and June. I collected data in the field in January, February, and March of 2024. This was primarily due to timing constraints. Collecting additional field data in May and June may offer greater confidence in plant identification, and possibly expand the coverage of field polygons.

In addition to challenges with the weather and hydrology, the areas near the grain silo, designated as built habitat, received treatment for HBB, primarily mowing, and habitat restoration is underway. Some of the mowed patches have been covered with thick mulch and replanted with various native shrubs and trees around the silo. The work accomplished between when the drone imagery was taken and when I conducted field research severely impacted the accuracy of field data polygons in a 10 square-acre section, approximately 9%, of the study area (Figure 12). However, the rest of the study area was not as impacted by restoration work. There were some other HBB patches that were mowed but are still visible because mulch has not been applied and about 6 inches of the canes remain, sticking out of the ground.

Figure 12

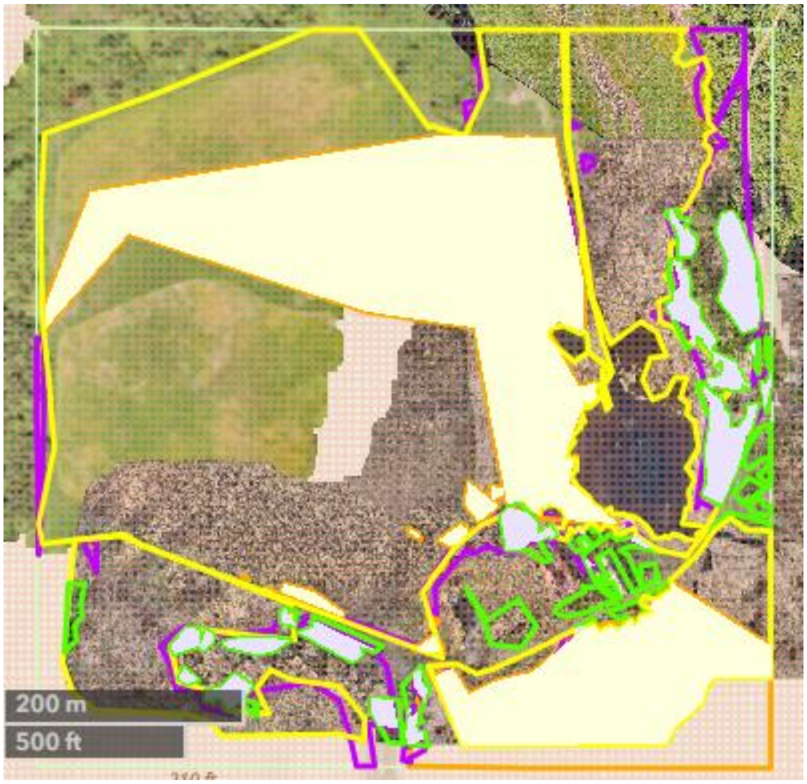
Study Area Impacted by 2023 Restoration Work

Note. Since this imagery was taken, the barns have been demolished, and most of the visible HBB patches around the barns (green polygons) have been mowed and covered in mulch. There are noticeably less purple polgyons, which were drawn on the ground.

Figure 13 shows the subsection of the map with the highest concentration of polygons, which measures out to about 90 square acres. Both layers of polygons are displayed, over the drone imagery orthomosaics, as well as computer-generated polygons, based on where the two sets of polygons intersect. There is a small gap in the middle and along the bottom of the grid where drone imagery was not collected, but the gap area is over a field, and low in variability. A light red grid of 45 square meter blocks, which is roughly equal to half an acre, marks the level of precision I applied, based on the needs of the restoration project manager. It was most important to identify whether either species was present in a half-acre section, but less important to get an exact measurement of how much. Getting a sense for where HBB or RCG has spread is a higher priority at this stage in the restoration work at the preserve.

Figure 13

Subsection of Study Site Used for Data Analysis



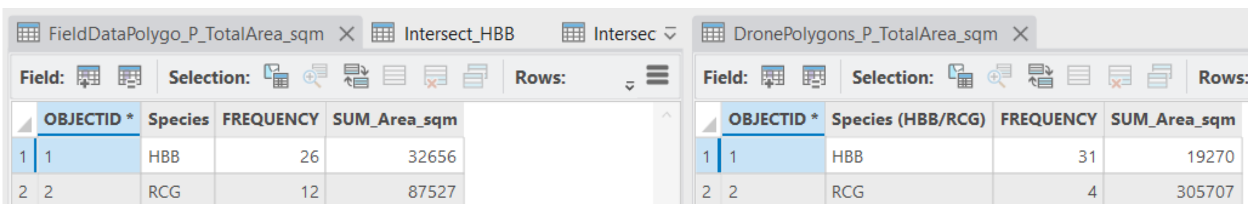
Note. Light green border highlights the study area, which included 90 acres of preserve, while the light red grid marks roughly half acres. Yellow polygons represent RCG and green polygons represent HBB based on drone imagery. From the field, orange polygons represent RCG, and purple polygons represent HBB. The intersections of drone HBB to field HBB polygons are light purple polygons, while the intersections of drone RCG to field RCG polygons are light yellow.

To focus the data analysis on the polygons within the designated subsection of the study area, I had to remove the extraneous polygon data. I used Generate Tessellation to make a grid of 45 square meter squares. After identifying which grid squares covered the area, I used Dissolve to combine the 9 grid squares, and created an outer boundary around the analysis area. Then, I used a Pairwise Clip on both polygon layers to remove polygons and areas of polygons that fell outside the boundary.

To compare the drone polygons to the field polygons, I first calculated the area of each polygon and all the polygons by species. Using Calculate Geometry, the software calculates the area of each polygon in the unit I designated, square meters. This allowed me to enrich the labels of the polygons with this information, as well as Summarize the new "area_sqm" field to generate a new table where the total combined area of each type of polygon by species is calculated (Figure 14).

Figure 14

"Summary Statistics" Geoprocessing of "Area_sqm" Fields



FieldDataPolygo_P_TotalArea_sqm

	OBJECTID *	Species	FREQUENCY	SUM_Area_sqm
1	1	HBB	26	32656
2	2	RCG	12	87527

DronePolygons_P_TotalArea_sqm

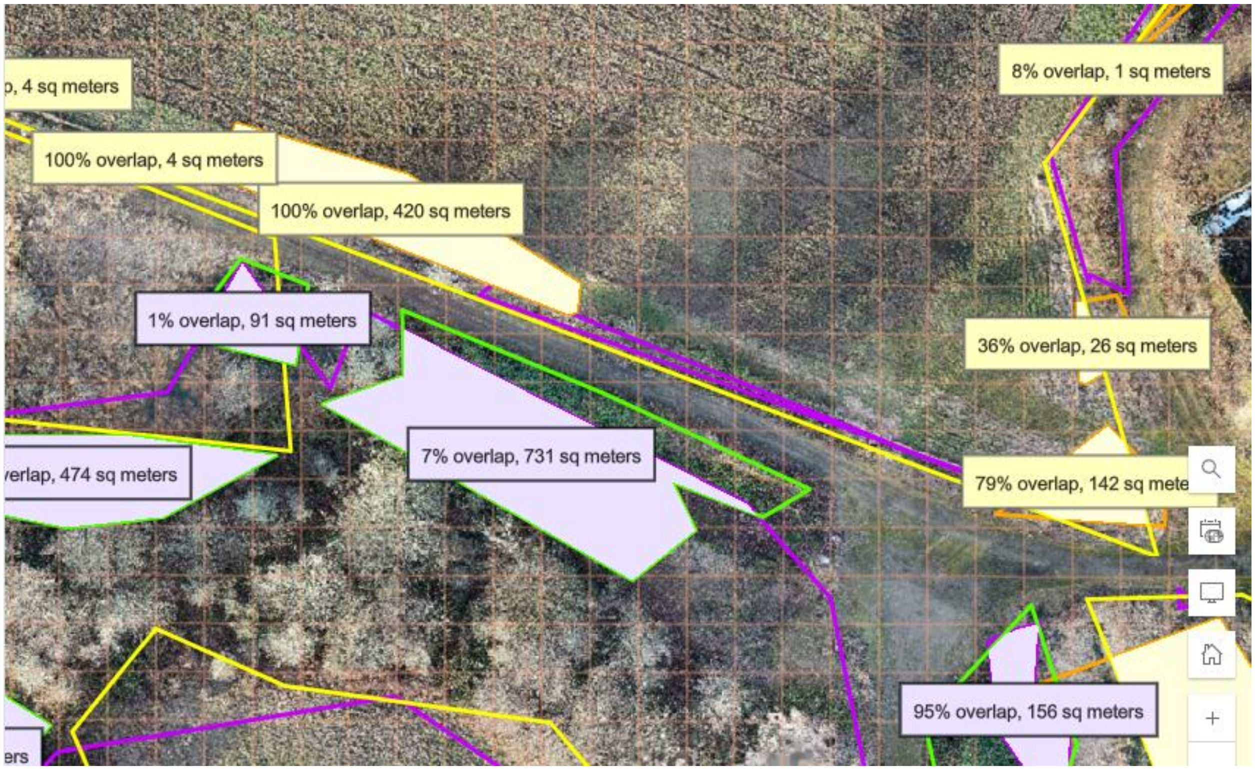
	OBJECTID *	Species (HBB/RCG)	FREQUENCY	SUM_Area_sqm
1	1	HBB	31	19270
2	2	RCG	4	305707

Note. The table on the left is a summary of the number and total area in square meters of Field polygons by species, and the table on the right displays the Drone polygons by species, within the subsection of the preserve used for this study.

The final geoprocessing tool applied was Intersect, which found the overlap of polygon layers, and produced a new layer of polygons based on where they intersected. I ran the Intersect tool on HBB polygons first, then RCG to create two separate layers of intersect polygons. In Figure 15, the HBB intersect polygons are light purple, and the RCG intersect polygons are light yellow. Again, I used Calculate Geometry to generate the areas of each intersect polygon, used Calculate Field to generate a ratio of overlap by dividing the intersect area by the total area of field polygons, and finally converted the ratio to a percentage by using Calculate Field once more to multiple the ratio by 100. The labeling of each polygon was updated to display the percentage of overlap based on field polygons, and total area of the intersect polygon.

Figure 15

Intersect Polygons with Labels



Note. Zoomed in to display labels and information generated by geoprocessing tools.

A limitation of using the Intersect tool to analyze this data is it only accounts for overlaps, and polygons that do not intersect will not be reflected in the results. The assumption is that field polygons are closer to the "truth" of what exists, so drone polygons that do not intersect become extraneous. Table 1 summarizes the statistics generated by the steps above.

Table 4

Summary of Statistics of Drone and Field Polygons by Species

	# of Drone Polygons	Area of Drone Polygons (sq m)	# of Field Polygons	Area of Field Polygons (sq m)	Area of Intersection (sq m)	Average Overlap
Himalayan blackberry	31	19,720	26	32,656	10,276	37%
Reed canary grass	4	305,707	12	87,527	76,424	82.25%

Note. This data is limited to the analysis area, not all of the polygons collected across the entire preserve.

To include the entire area of drone polygons, not just the areas of intersection, a percent change calculation comparing the total area of drone polygons to total area of field polygons would reflect a more inclusive percent difference:

$$\frac{total\ area\ of\ drone\ polygons - total\ area\ of\ field\ polygons}{total\ area\ of\ field\ polygons}\ x\ 100$$

HBB:

$$\frac{19720 - 32656}{32656}\ x\ 100 = -39\%$$

RCG:

$$\frac{305707 - 87527}{87527}\ x\ 100 = 249\%$$

For HBB, the total area of polygons derived from drone imagery ended up being less than the total area of polygons from field data and yielded a negative percent change of –39%. For RCG, it was the opposite, and the percent change was +249%. Using these percentages to analyze the difference between drone and field datasets, RCG was overestimated in drone imagery, while HBB was underestimated.

5. Discussion

5.1. Assessment of methods

Considering the stark difference in results between the two study species, manual interpretation of drone imagery can be used as a supplement to field data collection but is not suitable as the only method for assessing and monitoring invasive plant species. Overlap analysis indicates a high average overlap for RCG polygons (82%) but low average overlap for HBB polygons (37%). This means drone polygons were more similar positionally to the field polygons for RCG, but HBB polygons were in different locations. This is likely due to manual interpretation of drone imagery overestimating RCG presence but underestimating the presence of HBB. While accuracy is the goal, restoration managers are better served with overestimations, as underestimates could result in insufficient resources and funding. This study finds that applying manual interpretation of drone imagery in the specific case to quantify a highly visible and dominant invasive species, such as patches of RCG in fields or along waterways, greatly expedites the data collection process. However, in the case of understory plants like HBB, drone imagery is insufficient to accurately identify and measure abundance, especially in habitats with other understory plants and trees where visibility is obscured.

5.2. Method efficiency

Comparing the drone polygons to field polygons, and the time spent to analyze and collect data, I cannot confirm the efficiency of using remotely sensed data, even with coarse (half acre) accuracy requirements. The total time spent drawing polygons using drone imagery, including time spent calibrating methods was 7 hours, which is less than the 11 total hours I spent in the field. It is important to note that this time required for drone imagery analysis does not include time spent for multiple drone flights and processing the imagery into orthomosaics, which would likely be much more than 11 hours. When analyzing the drone imagery, it was difficult to discern both species in mixed trees/shrub areas, but highly efficient for field, wetland, marsh, and built areas. In the field, it was noted that some RCG and HBB had encroached into forested areas, but these polygons were either not captured or not included in the analysis. Due to time constraints, the scope of this study was limited to a section of the preserve, which ultimately did not include the densely forested sections.

5.3. Accuracy limitations of field data

Data collection on the ground was greatly affected by a higher volume of standing water present in the wetland areas during the winter season when this research was conducted, and thick understory growth, which inhibited foot travel through mixed trees and shrub areas. This development inhibited the scope of this study and impacted the amount of field data collected. It was also much more difficult to decide whether RCG was present in certain areas due to potential mowing, and seasonality which impacted the stage of growth. It is important to note there is drone imagery taken during different seasons between the various orthomosaic layers, and I conducted my field study during the “off-season” or winter. A large field that looks to be covered in RCG in the drone imagery was predominantly covered in dead thistle and other species during this study, and only the banks of a small stream in the field had clear RCG establishment. Seasonality and mowing both play a significant role in discerning between tall and short grass species.

6. Conclusion

Habitat restoration is an act of hope in the face of eco-anxiety and despair about the changing climate. However, managing invasive species is an expensive, and seemingly endless, process in highly disturbed areas surrounded by urban development. Restoration teams toil against the resistance of the problematic plants and the re-introduction of seeds which flow, fly, and fall from all directions. The concentration of human activity seems to be directly proportionate to the introduction and spread of invasive species across habitat types. Leveraging available remote sensing technology to improve assessment, planning, and application of invasive management is possible at various price points. Drone imagery has been successfully used to detect vegetation in multiple habitat types (Neyns & Canters, 2022, Dash et al., 2019). There is great potential to expedite and enable mapping for measuring and monitoring invasive species in wetlands and other difficult to access habitats.

This study explored the use of drone imagery to detect and identify two different invasive species across multiple habitat types. One of the study species, reed canary grass (*P. arundinacea*), grew along streams and at the edges of ponds, and was clearly identifiable through manual image interpretation. It was much easier to measure in the drone imagery than on the ground, as all field measurements were done on foot. The other study species, Himalayan blackberry (*R. armeniacus),* had grown dense patches that were discernable in the imagery, except in forest and mixed shrub habitats. The total area of polygons derived from drone imagery were compared to the total area of polygons from field data. Percent change calculations between drone and field datasets were a simple way to determine that RCG was overestimated through manual visual inspection (249%), while HBB was underestimated (-39%). Understanding the accuracy of this method helps assess the viability of replacing traditional field surveys, especially in wetlands and marshes where standing water can make access difficult or even dangerous.

While remote sensing of invasive plant species has rapidly evolved over the last two decades, further research and development are required to achieve detection accuracy that could replace surveying on the ground. Despite successful implementation in agricultural settings (Dash et al., 2019), remote sensing of invasives does not translate as well to other landscapes. In the context of restored habitats with greater vegetation heterogeneity (Larkin et al., 2016), detection is especially challenging. Studies find that remote sensing methods are useful for differentiating between trees and shrubs but fall short in identifying individual plants or specific species (Neyns & Canter, 2022). There has been some success when a species grows in monotypic stands or large, dense patches, such as giant hogweed (*Heracleum mantegazzianum L.)* (Hill et al, 2017). Depending on the species and habitat types, remote sensing, and especially drone imagery, can be used to generate rough estimates of how much of an invasive plant species is present, which may still be useful for some use cases.

Land managers attempting to quantify the extent of a known invasive plant may be able to utilize drone imagery to accomplish this more quickly. Visual analysis of the drone imagery for this study took less time than collecting data out in the field, in part because of the lack of precision required by the land manager. The presence of either invasive species within a half acre was more significant than the precision of the polygons to the land manager. This allowed for faster drawing of polygons, both in the manual interpretation and while collecting data in the field. ESRI's Field Maps application also made data collection in the field fast and easy. Having well-designed software greatly enabled both the drawing of the drone and field polygons for this study.

Drones are more readily available, less expensive, and more flexible than other remote sensing platforms. Rotary-wing drones are also able to hover, fly lower, and slow down which allows for higher resolution imagery and inspection. The ability to launch and operate a drone from a safe location reduces risks to crew and equipment, as well as reduces impacts on the ecosystem being mapped (Hill et al., 2017). That being said, for smaller restoration projects, it could be potentially faster to build a GIS-based data collected on the ground, especially if the restoration team lacks the technical skills to process drone imagery into a usable orthomosaic. In some cases, terrain or water may prevent ground measurement, and drone imagery may be used as a supplement. Building a useful map does not require remotely sensed data in the context of invasive plant species at smaller habitat restoration sites.

Mapping could be accomplished when applying treatments, replanting, and other restoration activities. Software such as ESRI’s Field Maps turn any smart device into a data collection tool, and with user-friendly design, training and implementation can be done in the field. Well-designed software does come at a price, but nonprofit discounts are often available. Once field crew are equipped and trained, mapping could be completed alongside other field work, saving time and generating the most reliable data for project managers to aid in prioritization and planning of restoration efforts.

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