University of Michigan

Herbivory, Climate Change and the Future Landscape of Isle Royale National Park: Developing an Herbivory Monitoring Program to Adaptively Manage the Park's Terrestrial and Aquatic Ecosystems

Project report for the partial fulfillment of an MS degree from the School for the Environment and Sustainability

Client: National Park Serve, United States Department of the Interior

Advisor: Inés Ibáñez

Jordon Tourville, Margaret Lindman, Charlotte Weinstein, Robin Schultze, Mirko Noack
4-17-2018

Abstract

Climate change is expected to play a major role in the restructuring of forests and other vegetation communities across the globe in the coming century. Forests in particular currently occupy almost half of the Earth's terrestrial surface, with other types of natural plant communities contributing a large proportion of land cover. Given the importance of these ecosystems with respect to their biological functions (i.e., water provisioning, climate regulation, nutrient cycling, carbon sequestration, habitat provisioning, etc.), understanding how these plant communities will respond to the growing threat of climate change will allow us to anticipate and quantify significant impacts to long-term human and environmental well-being. Climate change and subsequent rising temperatures, in concert with biotic drivers such as herbivory, is thought to have already altered the composition of plant communities in Isle Royale National Park (ISRO). Understanding what continuing changes these communities may still undergo is a major goal for National Park Service (NPS) resource managers, as this information will allow for more targeted conservation strategies. In this report we present a vegetation and herbivory monitoring protocol (Part I) intended to assess the combined effects of herbivory and climate change on the terrestrial plant communities of Isle Royale National Park (ISRO). This protocol will capture data on vegetation composition and structure, and on associated moose browsing of terrestrial (herbaceous, shrub, sapling, and overstory layers) vegetation to identify areas within the park most susceptible to forest change. We also present recommendations for the development of a protocol to monitor aquatic vegetation. The information collected through future applications of this protocol will inform park managers which plants are most vulnerable to heavy browsing, where vegetation in the park is expected to change in the future, and the effect that variable climate and wolf predation on moose will have on the vegetation of ISRO. In addition, to address NPS needs, our team has conducted a review of the available literature concerning direct and indirect climate change impacts on the vegetation communities of ISRO (Part II). Using the classification scheme of The Nature Conservancy (TNC) and the United States Geological Survey (USGS), we described 14 unique ecological communities on ISRO, including the dominant plant species found in each community. Based on our literature review of both community- and species-level climate change impacts in this system, we present a conceptual framework for how different ecological vegetation communities and the dominant species within them may respond to the multifaceted effects of climate change on ISRO in the coming century. Our review identifies potentially "winning" and "losing" vegetative community types under commonly predicted climate change scenarios in the upper Midwest region. This synthesis of literature provides resource managers a first pass at estimating climate change effects on the plant communities of ISRO and will help in prioritizing areas for climate change mitigation.

Part I: Moose Herbivory Monitoring Protocol for Isle Royale National Park

National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science

Moose Herbivory Monitoring Protocol for Isle Royale National Park

Natural Resource Report

ON THIS PAGE
Rocky shoreline of Isle Royale on Mott Island
Photo taken by J. Tourville

ON THE COVER
Female moose with offspring browsing in wet meadow
Photo taken by J. Tourville

Moose Herbivory Monitoring Protocol for Isle Royale National Park

Natural Resource Report

Jordon Tourville[1], Margaret Lindman[1], Charlotte Weinstein[1], Robin Schultze[1], Mirko Noack[1]

[1]University of Michigan, School for Environment and Sustainability
440 Church Street
Ann Arbor, Michigan 48109

April 2018

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

Please cite this publication as:

Tourville, J., C. Lindman, M., Weinstein, C., Schultze, R., and Noack, M. 2018. Moose Herbivory Monitoring Protocol for Isle Royale National Park. Natural Resource Report. National Park Service, Fort Collins, Colorado.

Contents

Figures

Page

Tables

Page

Appendices

Page

Executive Summary

In this report we present a vegetation and herbivory monitoring protocol intended to assess the combined effects of herbivory and climate change on the terrestrial plant communities of Isle Royale National Park (ISRO). This protocol will capture data on vegetation composition and structure, and on associated moose browsing of terrestrial (herbaceous, shrub, sapling, and overstory layers) vegetation to identify areas within the park most susceptible to forest change. We also present recommendations for the development of a protocol to monitor aquatic vegetation. The methodology employed includes metrics to measure and monitor plant communities combined with methods to quantify moose herbivory and available forage. This protocol was developed in order to facilitate the collection of long-term data that expose the ways in which both herbivory and climate change might interact to alter the ecosystems of Isle Royale in the coming decades. Collected data will be taken from across the park, cover most vegetation communities, and will be collected on an annual basis. The information collected through future applications of this protocol will inform park managers which plants are most vulnerable to heavy browsing, where vegetation in the park is expected to change in the future, and the effect that variable climate and wolf predation on moose will have on the vegetation of ISRO.

View of ISRO central ridgeline as seen from Mt. Ojibway (photo credit: J. Tourville).

Acknowledgments

The authors of this report would like to thank NPS employees Lynette Potvin and Mark Romanski for their overwhelming support and guidance for this project. We would also like to thank other Natural Resources staff for their logistical support during the data collection phase of this work. We would like to also recognize Dr. Inés Ibáñez and the Global Change Ecology Lab at the University of Michigan for helpful insight with protocol design, report writing, and data analysis. We would like to extend our gratitude to Kirk Acharya and Douglas Levitt for their assistance in our initial scouting trip to Isle Royale. Lastly, we would like to recognize both the School for Environment and Sustainability, and the University of Michigan's Matthaei Botanical Gardens and Nichols Arboretum for their generous funding which helped support the work of this project.

List of Terms

NPS	National Park Service
ISRO	Isle Royale National Park
ABI	Accumulated Browse Index
DBH	Diameter at Breast Height
GLKN	Great Lakes Network Inventory and Monitoring Program
NDVI	Normalized Difference Vegetation Index
GRTS	Generalized Random Tessellation Stratified

Introduction

Statement of Task

The overall goal for this project is to provide the means of collecting the necessary data to predict how both herbivory and climate change will interact to alter the forests of Isle Royale in the coming decades. Climate change is expected to shift the distribution and composition of many forests in the coming century, in addition, biotic interactions, such as herbivory by large ungulates (moose), can strongly influence changes in vegetation. To address the combined effects of herbivory and climate change on the composition of forests on Isle Royale, the National Park Service (NPS) has tasked our team with the creation of a long term monitoring protocol assessing the effects of moose browsing on the vegetation. Data collected through the implementation of this protocol will be used to identify areas and species within the park that are most susceptible to vegetation change. A pilot protocol was tested in July 2017 to determine the feasibility of an island-wide monitoring protocol, to collect initial data, and to explore possible analyses. We have developed a protocol of annual data collection, and provided recommendations for the NPS to utilize for their monitoring and data analysis activities.

Isle Royale

Isle Royale was established as a National Park in 1940, and has since been only marginally impacted by anthropogenic activities (Flakne 2003). The park is famous for the moose herbivory and wolf predation studies that have been taking place for several decades (Vucetich et al. 2002; Vucetich and Peterson 2004; Nelson et al. 2008). In addition, much of the forested land on Isle Royale is located on the transition zone between the southern extent of the boreal forest and far northern extent of the temperate deciduous forest (Fisichelli et al. 2012). As a result, key tree species of this ecosystem could be vulnerable to both increased moose browsing and a warming climate. Understanding the combined effects of herbivory and climate change on the vegetation will then be critical to develop any conservation and management plans for the Park.

Isle Royale Natural History

Isle Royale National Park (ISRO) is a federally listed National Park Service (NPS) unit composed of an archipelago of approximately 450 islands in Lake Superior, of which 99% is managed as wilderness area (Sanders and Grochowski 2013) (Figure 1). The archipelago is dominated by one main island approximately 72 km long and 14 km wide. The majority of this island is forested, with boreal spruce-fir forests encompassing the northern end and shoreline, and northern hardwood forests covering the southern end (Sanders and Grochowski 2013) (Figure 2). The distribution of forests on the main island is determined largely by seasonal temperatures and soils (Kraft et al. 2010). The mean annual temperature is 4.9°C, the mean January temperature is -9.7°C, and the mean July temperature is 18.8°C. The position of ISRO within a large water body like Lake Superior makes it subject to lake-effect temperature modulation, with short summers slightly cooler than the mainland, and long cold winters slightly warmer than the mainland. Fog and heavy precipitation is common for ISRO year-round, with mean yearly precipitation of 75 cm (Kraft et al. 2010). The areas covered by spruce-fir forests, mostly at the lowest elevations, are colder and moister on average throughout the year, with minimum winter temperatures limiting the growth of other types of forest communities in these areas (Bonan and Shugart 1989; Brandner et al. 1990; Pastor et al. 1998). The island's soils also constrain the distribution of certain forest types. Deeper and more developed alluvial soils on the southern end of the island support deciduous northern hardwood forests typical of mesic sites, while the northern end of the island is characterized by thin, dry, young, and poorly developed soils and exposed volcanic bedrock (Kraft et al. 2010). The unique geologic history of the island also plays a role in forest distribution. The island is composed of a series of parallel bedrock plates running along a southeast axis. The uplift of the island post-glaciation resulted in a topography of parallel ridges and valleys, reaching a maximum height along the central ridge that runs the length of the island. The intervening valleys collect water, and support black ash and alder swamps, as well as wet meadows and inland lakes (Kraft et al. 2010).

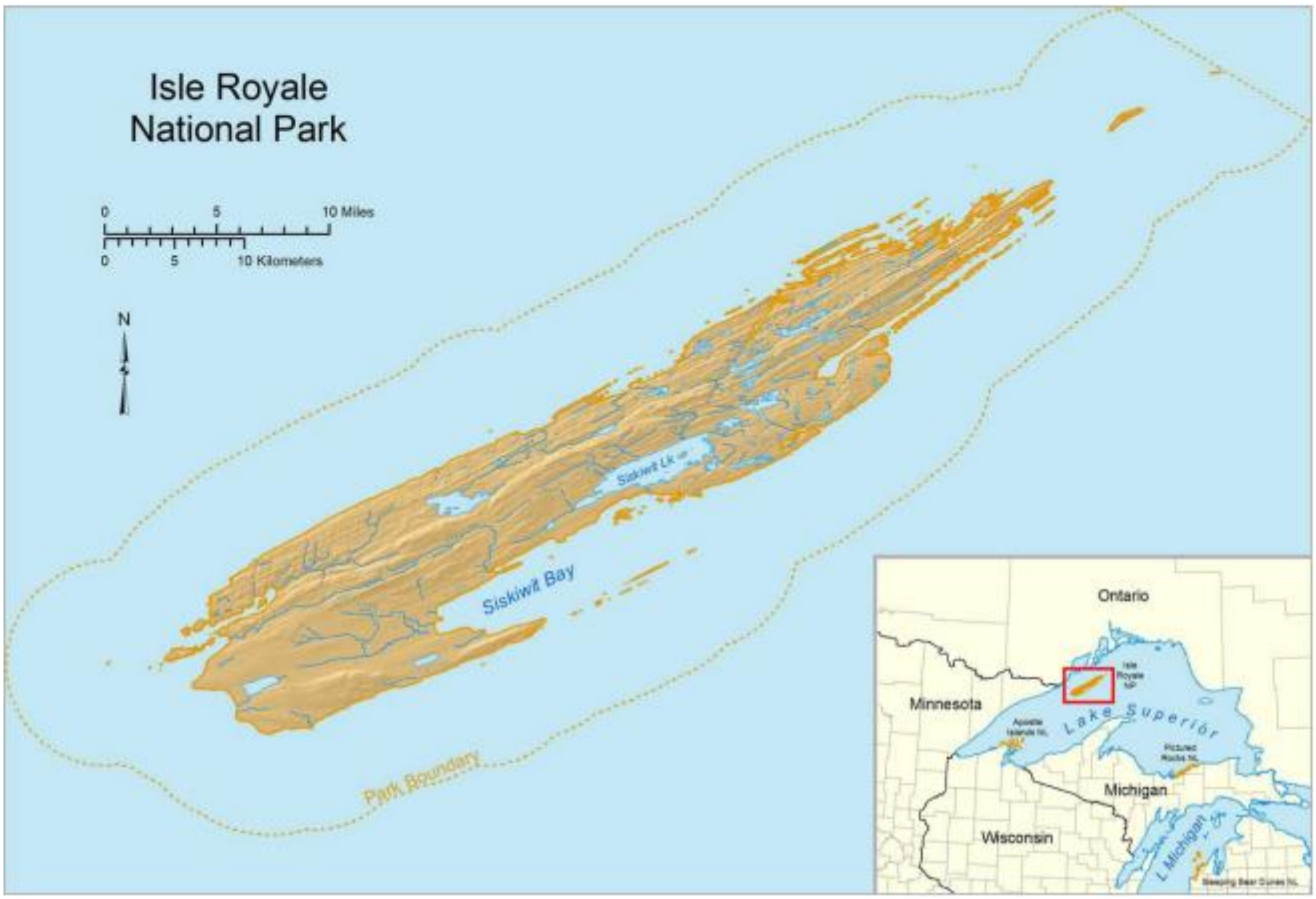


Figure 1. General map of Isle Royale National Park (taken from Kraft et al. 2010).

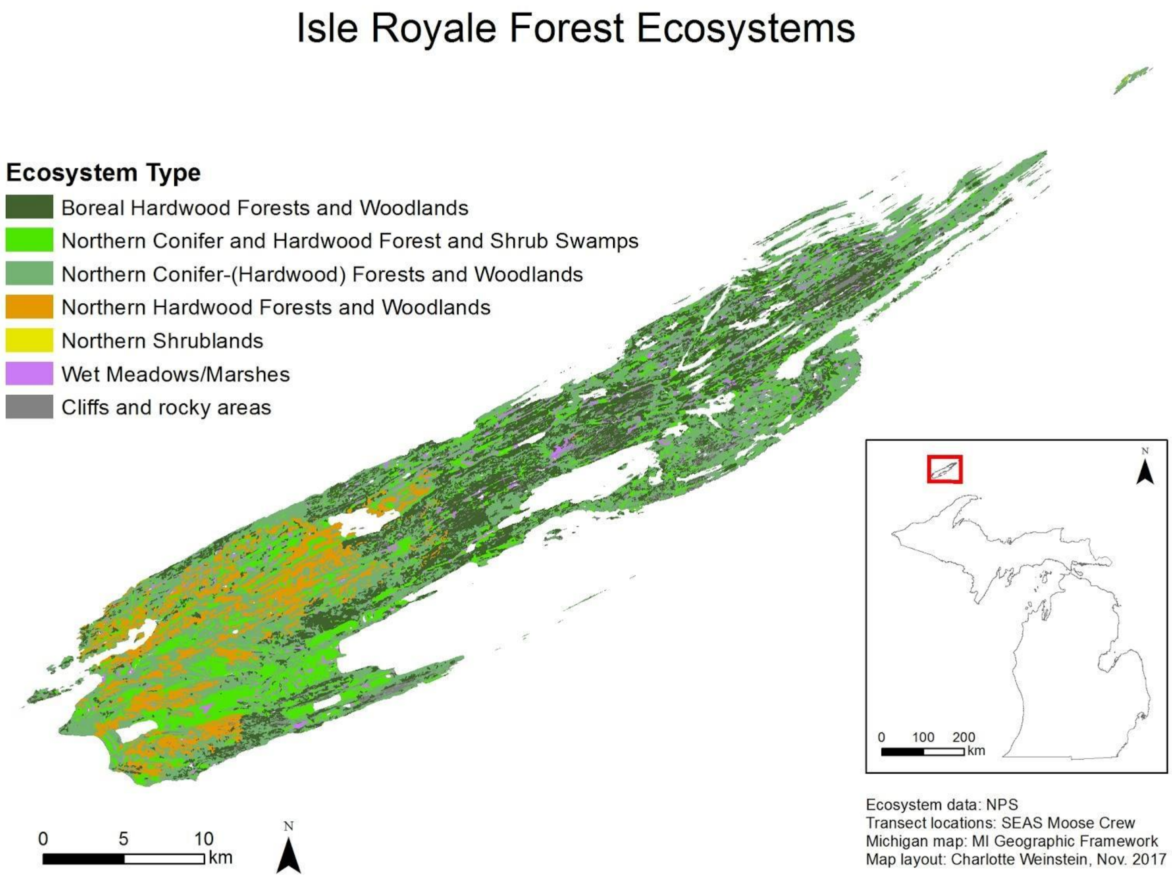


Figure 2. Map of common forest community types found on ISRO.

ISRO has many inland lakes, perennial and intermittent streams, and wetlands. There are 202 inland lakes across the park ranging in size from 1,635 ha (Siskiwit Lake), to numerous small and shallow ponds (Kraft et al. 2010). The lakes are a product of the unique topography of the island, coupled with glacial action. Many of the lakes are long and narrow; their shapes defined by the southwest to northeast ridge and valley orientation of the main island. Numerous studies cataloging aquatic taxa have been conducted in ISRO. In particular, studies highlighting aquatic macrophyte diversity (both emergent and submerged) have encountered over 400 species (Meeker et al. 2007).

Drivers of Change in ISRO Ecosystems

Climate Change

In their report "Using Climate Change Scenarios to Explore Management at Isle Royale National Park", Fisichelli et al. (2013) identified and explored four plausible climate trajectories for Isle Royale through 2050. Scenarios ranged from 'Least Change,' defined by a mean annual temperature increase of 3.4 °F and an annual 5% increase in precipitation, to more extreme scenarios including 'Summer Drought, Wind, and Fire' characterized by drier summers and more severe winds, the 'Warmer than Duluth' scenario with temperature increases of 6.5 °F, and the 'Isle Savanna' scenario with temperature increases of 5 °F accompanied by greater variability in precipitation (10%-15% increase) and increased drought. Thus, the three extreme scenarios generally correspond with intensified and more frequent disturbance events. These scenarios will most likely cause dramatic changes to the forest communities in ISRO. Figure 3 displays likely changes to both plant communities and major keystone mammalian species under the four proposed climate change scenarios. In all cases, boreal tree species are projected to decrease in abundance and distribution, while temperate trees species (such as sugar maple and yellow birch) are likely to increase in abundance and distribution in three of four scenarios. Thermal and water stress associated with increasing temperatures and an altered precipitation regime may limit the potential range of boreal species, which at present are strongly tied to cool and moist areas of the main island (Allen et al. 2010; Sanders and Grochowski 2013; Evans and Brown 2017). Increased interspecific competition between more thermally adapted temperate deciduous species and stressed boreal species may also play a role in the reshaping of forest communities on ISRO. Moreover, these temperate species may be able to better cope with increases in disturbance frequency than their boreal counterparts (Dale et al. 2001; Goldblum and Rigg 2010; Fisichelli et al. 2014). However, heavy browse can also limit the expansion of palatable deciduous species, and the permanence of balsam fir (*Abies balsamea*), potentially resulting in an increase of savanna-like systems (Rotter and Rebertus 2014).

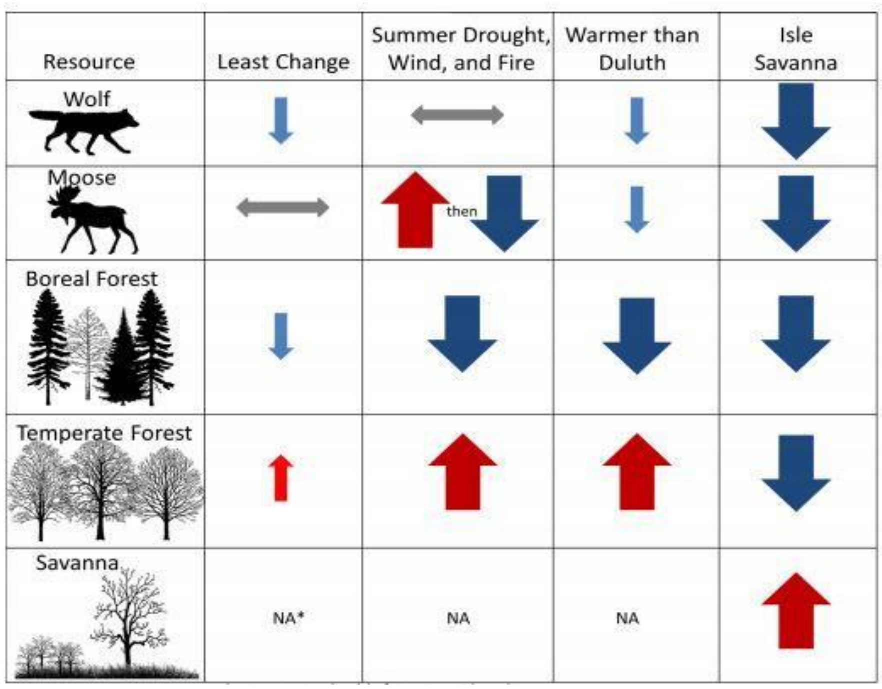


Figure 3. Illustration of predicted changes to ISRO forest communities and key mammals under four proposed 2050 climate change scenarios. Taken from Fisichelli et al. (2013).

Herbivory and Moose-Wolf Interactions

Wildlife populations and predator-prey dynamics on ISRO have fluctuated widely during the past century. There are no records of either wolves or moose inhabiting Isle Royale prior to 1900; while small moose populations might have existed, it is thought that the first large influx of moose occurred during the winter of 1912-1913 (Murie 1934). Following this, the moose population on Isle Royale rapidly increased to 3,000 individuals by the 1930s. The increase brought starvation and a resulting population crash in 1934; however, the population seemed to recover by the late 1940's (Peterson et al. 1998). This is a common dynamic for moose populations that often follow a boom-and-bust cycle: increases in their population result in an overgrazing of the landscape, which in turn leads to a decline in the overall moose population as they struggle to find new sources of nourishment (Nelson et al. 2008).

Figure 4. Wolf numbers on ISRO have crashed in the last decade (photo credit: R. Peterson).

Wolves were first observed near Isle Royale in the 1940s, reaching a peak population of around 50 individuals in the late 1970's. Between 1980 and 1982 the population declined from around 50 to 14 (Peterson et al. 1998). This population crash is thought to have been caused by genetic decay from isolation, food shortage, new disease, and demographic stochasticity (Peterson et al. 1998). As of 2016, the number has dwindled to only two individuals (Figure 4). This is important to note, as wolves are the only predators for moose on the island and contribute to the stability on the ecosystem as a whole. The severe decline in wolf numbers has caused burgeoning of the moose population, now estimated at over 1,600 individuals (Figure 5) (densities in some areas of 6 moose per km^2), which could have cascading effects on vegetation on the island (Figure 6) (Vucetich and Peterson 2017). As a consequence, ISRO is a very unique place to study moose browse effects on vegetation, as the lack of predation has created an artificial system with moose densities much higher than anywhere else on the continent.

Figure 5. Moose numbers have been increasing as a result of relaxed predation (photo credit: L. Potvin, NPS).

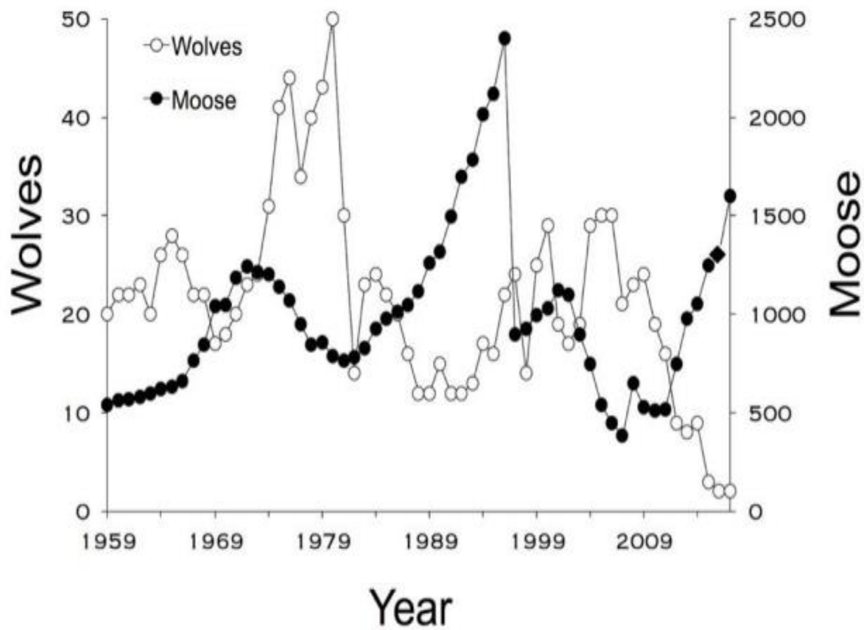


Figure 6. Moose and wolf population dynamics have fluctuated over the five decade long study period (figure taken from Vucetich and Peterson (2017)).

Plant animal interactions in the context of climate change

At the latitudinal limits of both temperate and boreal species' ranges, combined pressures of a warming climate and moose browse will inform the diversity and relative dominance of forest types on Isle Royale. While temperate species are expected to proliferate with a warming climate and boreal species are expected to shift north, in some instances, warming and browse may serve as opposing pressures, such that intense browsing counteracts increased growth of temperate broadleaf species and facilitates the relative dominance of boreal species. That said, moose are also at the southern boundary of their distribution, and warming could negate their effects entirely (Fisichelli et al. 2012; Sanders and Grochowski 2013). The complexity in predicting the outcome of these pressures is heightened by the temporal mismatch between predator-prey dynamics and climatic shifts, as well as the growth rates of species that compose these future forests (McLaren 1996; De Jager et al. 2017). This dynamic reiterates the need for long-term monitoring that tracks browse-induced vegetation shifts to more accurately predict the structure and function of landscapes under climate change scenarios (Zarnetske et al. 2012; Gebreyohanis 2017).

Heavy browsing may result in vastly different, or in some cases, non-analog forest communities (Rotter and Rebertus 2014). Moose browse preferentially on the leaves and twigs of deciduous plants during the summer, and utilize more coniferous plants during the winter months (primarily balsam fir) (McLaren 1996). However, the effects of moose on the island are myriad: the annual rubbing of velvet from their antlers has the ability to girdle saplings and small trees, and when browsing saplings, it is not uncommon for moose to snap or tear them down completely (Murie 1934). As a result, forests formerly dominated by highly preferred species will be represented to a greater extent by species that are less palatable and less competitive. This will in turn lead to changes in litter quality, differing interactions between plants and other consumers, and altered conditions for plants in the understory.

Ungulate herbivores like moose are major players in ecosystem level processes including nutrient cycling and forest succession (Frerker et al. 2013). A number of studies have suggested that herbivory by large ungulates can greatly influence forest response to climate warming by reducing the competitive dominance of deciduous trees and shrubs (Post and Pederson 2008; Fisichelli et al. 2012). Deciduous trees and shrubs are more palatable and have higher nutrient content (N and P) than common co-occurring boreal woody species and graminoids (Evans and Brown 2017). Thus, these species can be highly favored by ungulate grazers and as a result the competitive edge conferred on deciduous species under a warmer climate could be counterbalanced by increased herbivore pressure on the leading edge of their range.

Modulation of forest expansion into novel ranges under a warming climate by ungulate herbivores may have implications for ecosystem level processes like carbon and nutrient cycling. Forest species differ in the quality and quantity of their litter, as well as their rates of net carbon uptake from the atmosphere (McInnes et al. 1992; Pastor et al. 1993, 1998; Chapin et al. 2008). As tree species composition changes in response to herbivory and warming the rates of carbon accumulation and nutrient cycling can dramatically change (Post and Pedersen 2008).

Other Threats

In addition to climate change and high herbivory pressure, the forests on ISRO face other major threats. Both exotic pests and plant species have the potential to exclude native plants of conservation significance. Spruce species are susceptible to the spruce budworm which has the potential to eliminate large patches of standing forest and promote increased fire frequency (Dale et al. 2001). Another exotic pest not currently present on ISRO is the emerald ash borer. This insect targets all ash species, which includes the black ash that is very common in wet areas of ISRO (Kraft et al. 2010). In addition to exotic insect pests, invasive plants, such as invasive cattails and the common reed have the potential to dominate inland aquatic systems, potentially altering the dynamics of these diverse systems (Kraft et al. 2010).

Climate change may also promote the establishment and spread of invasive species on ISRO in the future. An increase in disturbance frequency (windthrow and fire) creates conditions ideal for early successional species, like many invasives, to establish and suppress the recruitment of native plant species (Dale et al. 2001). This may be relevant to moose browse dynamics, as the spatial distribution of invasive species on the island may alter moose foraging behavior. Monitoring, prevention measures, and prompt spot treatment of exotic plant and insect outbreaks will be critical moving forward for ISRO resource managers.

Previous Browse Monitoring

The first major efforts to assess the impact of moose and moose browsing on ISRO were conducted in the 1940s with the construction of five permanent 50 x 50 feet moose exclosures (Kraft et al. 2010).

Over several decades, notable vegetation differences were observed on both sides of the exclosure fence, which include the near extirpation of Canada yew (*Taxus canadensis*) and the decline of trembling aspen (*Populus tremuloides*) in areas exposed to moose herbivory (Risenhoover and Maass 1987; Pastor et al. 1993; Sanders and Grochowski 2013). In terrestrial systems, balsam fir (*Abies balsamea*), trembling aspen, paper birch (*Betula papyrifera*), and some tall shrubs (e.g. *Corylus* spp.) are preferred moose forage on ISRO (Murie 1934; Janke 1978).

Based on surveys and aerial photography studying moose density and population size on ISRO for several decades, Rolf Peterson's research group concluded that greater moose densities are found in areas exclusively associated with the distribution of the boreal forest (Pastor and Naiman 1992; McLaren and Peterson 1994). In addition, Peterson's group were among the first to directly examine moose effects on aquatic vegetation using GIS based methods, noting a decrease in watershield cover in select inland lakes given higher moose densities (Peterson, unpublished data). These studies were able to pinpoint the types of forage generally preferred by moose throughout the year, and noted some variance in the intensity of browse across the main island. In particular, they reported high impact browsing on balsam fir on the southwest end of ISRO, especially after the release from predation pressure from decreasing wolf numbers (Vucetich et al. 2002; Vucetich and Peterson 2004). Other research has suggested that balsam fir browse intensity is determined to a greater extent by conspecific sapling and tree density, and that as a result, moose browse of balsam fir is very limited on the boreal northeast end of the main island (Brandner et al. 1990).

There have been two other park-wide efforts to quantify moose herbivory on ISRO. The first and longest running effort was conducted by Peter Jordan and his lab from the University of Minnesota. Jordan's field methods were first initiated in 1963, and were conducted annually for nearly 50 years. In total, 631 sampling plots were established along a longitudinal gradient, with plots separated by latitude (north to south axis) (Jordan et al. 2000). Each circular sampling plot was marked with a permanent stake. Within each circle, researchers measured tree species composition, moose and snowshoe hare density via fecal pellet counts, browsing as quantified by number of bite marks per plant, site characteristics (soils, hydrology, etc.), number of downed trees, and evidence of spruce budworm outbreak. The protocol outlined by the Jordan research team has the advantage of sampling the same sites through time; making predictions of forest composition change possible. However, repeated sampling of such widely dispersed points would be difficult given limited park staff and resources.

The second effort to quantify moose herbivory on a park-wide scale is through the Great Lakes Monitoring Network (GLKN) Long Term Vegetation Monitoring Program (Sanders and Grochowski 2014). The purpose of this effort is to monitor the change of forest composition and health through time given the changes in abiotic conditions projected under climate change. In ISRO, 52 permanent plots were randomly placed using the GRTS method (Stevens and Olsen 2004). The GRTS is an algorithm based method that randomly selects survey points within user defined regions, but stratifies, or weights, the number of points assigned in each region. The plots were first surveyed in 2010 and will be re-surveyed on a nine-year return interval. Each sampling plot utilizes the Hybrid plot design created by the GLKN and measures vegetation characteristics such as growth (DBH) of trees and saplings, amount of woody debris, presence of exotic earthworms, and herbaceous plant and seedling cover. Ungulate browsing is assessed in two ways. First, presence/absence browse of woody species within 68 1m x 1m browse plots within the Hybrid plot is tabulated, and the data are used to calculate the "browsing index" created by Morellet et al. (2001). This index provides region specific data on forage preference by analyzing binary browse data on individual woody plant species. Thus, the input data for this index are derived rapidly in the field. Second, browse is quantified via an indirect browse index comparing the relationship between individual herbaceous species abundance and the intensity of browse on co-occurring woody vegetation, as described by Frerker et al. (2013). In this way, the presence (or absence) of certain herbaceous plants within a plot can serve as an indicator for moose browse intensity. The methods presented by the GLKN provided an avenue for rapid field assessment of moose browsing and concurrent data collection on the status of the surrounding forest. While practical in design, the GLKN protocol did not accumulate much moose browse data from the 2010 survey. One of the main issues with these browse methods is that balsam fir did not occur enough in browse circles to be included in analysis, even though it is known to be a highly preferred browse species. This may have been the result of plot placement design, which while stratified to increase sample power to detect differences in forest type and vegetation, does not capture areas of high browse intensity. We suggest that the GLKN plots will provide extremely useful data on baseline vegetation metrics including regeneration of preferred and non-preferred browse species, but that a park-wide moose herbivory monitoring protocol is needed to complement these data.

Objectives of Proposed Protocol

- Given the potential effect of climate change on the vegetation of ISRO, and the need to understand how moose herbivory would shape the impact of these effects, we have sought to create a combined vegetation and moose herbivory monitoring protocol to track vegetation changes as a function of both climate change and moose herbivory. While snowshoe hare and beaver are also significant herbivores on ISRO, moose are the only large-bodied mammalian herbivore that occurs in the park and are the primary focus of this protocol. The ultimate goal of the proposed protocol is to estimate the ways in which climate change interplays with the impact of moose browse on forest structure and composition in Isle Royale National Park. The protocol also aims to be lead exclusively by ISRO park staff, be time-efficient, and be standardized through many field seasons. A pilot protocol was tested in July, 2017 and initial vegetation data collection and synthesis was conducted to formally develop the long term monitoring protocol, and to explore which research questions could be formulated with such data.

We have developed a protocol designed to:

- Assess the browse intensity, forage availability, and relative abundance and composition of forest plant species and communities on ISRO.

Methods

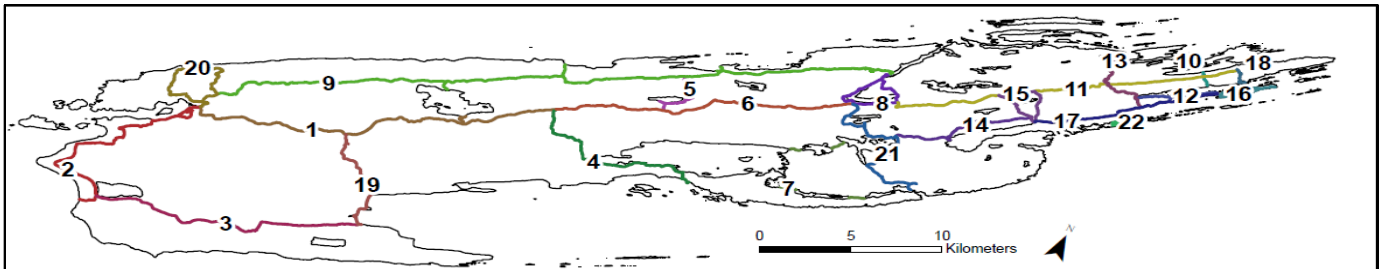
Transect Placement, Sampling Frequency, and Transect Design

The sampling protocol will follow the existing trail network of Isle Royale and be conducted once every year starting the last week in June and ending once all recommended data has been collected (most likely the end of July or the beginning of August). Sampling will be systematic, with each section of the trail containing a specified starting point and a fixed distance interval at which to set transects. Each point assessed on the trail will have two transects in that area. The sampling intervals for each trail will differ in time and space with moose population density and heterogeneity of vegetation. Areas with higher moose density (as determined by overlaying a digitized moose density map (Vucetich and Peterson 2017) with the trail network) will be sampled at shorter time intervals, e.g., annually, while areas with low moose density will be sampled at longer time intervals, every 3-5 years. In addition, in areas with higher vegetation heterogeneity (as determined by overlaying the 2000 TNC land cover layer on the trail network), e.g., sharp topographic and edaphic gradients, sampling locations will be set at smaller distances, e.g., 0.5 km, while in areas with more homogeneous vegetation, samples points will be further apart, e.g., 2 km. The trail network will be divided into 22 different sections, with the majority of sections including only one main trail with a unique name known to NPS staff. Appendix C, Table 1 describes the spatial sampling intervals, and sampling frequency for each of these sections (Figure 7).

For each point assessed along the trail network, two 30-meter transects will be established, 30 meters apart (Figure 5). A GPS point will be taken from the 0 meter mark of the first and second transect. The first transect will be placed on the right side of the trail when facing the walking direction, and the second transect will be placed on the opposite side of the trail (Figure 8), except if an obstacle or other exclusion condition is encountered (see Contingency Plan). The transects will begin 5 meters from the edge of the trail to compensate for any observed trail effect (i.e. higher proportional browsing closer to the trail), and will be placed in the cardinal direction closest to perpendicular to the trail.

Each transect will be denoted by the abbreviated trail name where the transect is found, the number of the point surveyed from the beginning of the trail, and given an A if it is the first transect on the point, or a B if it is the second transect uptrail of the first in the point. For example, if surveying the first transect of the first point on Feldtmann Lake Trail, the transect ID would be FLT1A. The second transect (transect uptrail of the central transect when facing starting point from trail) will be called B and the central (first) transect will be called A.

Based on observations from a pilot data collection survey conducted in July 2017, we recommend that 120 transects (60 sampling points x 2 transects per point) at minimum are needed annually in order to detect trends in browsing and vegetation change. We recommend that the NPS sample at least three times this number of points on a 3 year interval, and twice this number in the intervening years (resulting in approximately 360 transects every three years, and 240 transects in other years). During the pilot study we were able to complete each transect in 30 to 60 minutes. We anticipate that the proposed number of transects (360) will be possible to be surveyed in between 3-6 weeks (dependent on park resources and priorities). Since 2018 will be the first year of data collection, approximately 360 transects will be surveyed in 2018.



1, Greenstone Ridge Trail South (Windigo to Ishpeming Point)
2, Feldtmann Lake Trail
3, Feldtmann Ridge Trail
4, Ishpeming Trail
5, Hatchet Lake Trail
6, Greenstone Ridge Trail Central (Ishpeming Point to West Chickenbone)
7, Central Lakes Portage Trails
8, East Chickenbone Loop Trail
9, Minong Ridge Trail
10, Duncan Bay-Tobin Harbor Portage Trail
11, Greenstone Ridge Trail North (West Chickenbone to Monument Rock)
12, Tobin Harbor Trail
13, Mt. Franklin and Lane Cove Trail
14, Lake Richie Trail
15, Mt. Ojibway - Daisy Farm Trail Loop
16, Stoll Trail
17, Rock Harbor Trail
18, Lookout Louise Trail
19, Island Mine Trail
20, Huginnin Loop
21, Indian Portage Trail
22, Mott Island Loop

Figure 7. Map of ISRO trail network. Refer to Appendix C, Tables 1 and 2 for specifics on sampling set-up and schedule.

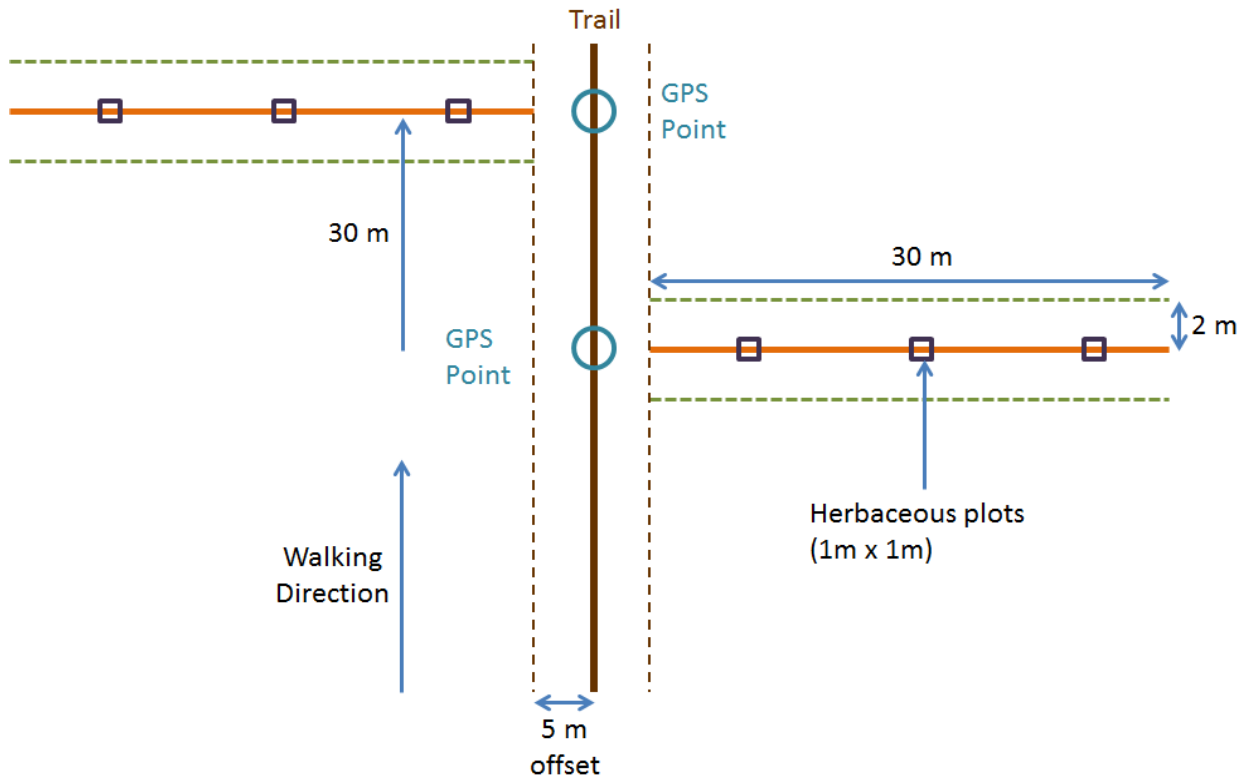


Figure 8. Illustration of the transect design (not to scale).

Contingency Transect Placement and Sampling

In addition to the official protocol recommendations, we have also developed differing sampling methodologies in order to better suit the varying time, resource, and management needs of National Park Service. In a limited resource scenario, we have outlined a possible abbreviated monitoring protocol:

- Reduce number of sampled transects by 50% while maintaining plot selection as described above. Remove trail sections with primarily heterogeneous vegetation first. Sample remaining transects during the following field season.
- Measure only one central transect at each sampling point, rather than two. It is possible for points to be located in areas that are inaccessible or do not adequately represent the variable of interest. In order to compensate for such instances, we describe a set of selection criteria to determine whether a sampling point will be measured, along with contingency steps in the event that a sampling point is deemed infeasible (see next section).

General Transect Placement and Sampling

Below are the criteria and alternate transect placement methods for the general transect placement:

- Individual points will first be assessed by technicians in the field to ensure points are not placed on exposed rock or water. The observed dominant canopy type must match what is described in the 2000 LULC map (TNC 1999; USGS 2000). We will not include a minimum canopy cover rule, as heavy moose browsing may occur in or result in an open savanna-like system; however, exposed rock outcroppings or areas lacking any vegetation will not be measured.

- In addition, points (and any part of the transects derived from them) must not be located in deep water (over 1 meter), or in unsafe areas, such as cliffs or extremely steep slopes.
- If the points, or the entire length of any of the resulting transects surveyed do not meet the above criteria, staff will move a transect 200 meters in either direction along the trail. If a suitable location cannot be obtained within this window, the point will not be used in the survey.

Personnel Training and Requirements

This field protocol requires 2 people per team (1-2 teams should be employed to carry out data collection). Depending upon the resources available to the park, these technicians can be employed for the entire field season, and assist the Natural Resources staff with other projects, or be employed only for the duration of protocol sampling (3-6 weeks). Preferred skills include plant identification (both woody and herbaceous) and experience laying out transects and plant biomass measurement techniques, such as measuring DBH and plant height. Due to the overall simplicity of the protocol, only 2-3 days of training should be necessary. This training period may include an overview of the field procedure, i.e., the layout of sample transects, orientation to the topography of the park, and a review of plant species found within the study areas. An overview of park-specific herbaceous plant identification would be a helpful component of this training. Field based GPS using ArcCollector or other phone-based applications may also be beneficial. Most importantly, technicians must become familiar with the herbivory quantification techniques used in this protocol: the bites taken/bites available ratio and ABI (defined in Herbivory Quantification section).

Potential Schedule for Personnel Training

Day 1

- Introduction to Isle Royale and forest communities
- Overview of protocol and its objectives
- Plant ID training: woody species and herbaceous species

Days 2-3

- Plant ID training: woody/herbaceous species
- Practice using herbivory quantification metrics
- Practice setting up transects and collecting data
- Practice with GPS units
- Practice setting up and collecting data along transects on Mott Island

Equipment

Recommended materials for data collection:

- 1-2 30-m transect tapes
- 2-3 DBH tapes
- Meter stick (to measure shrub and small sapling heights)

- 1 GPS using NAD 1983 datum (or one mobile device equipped with an application for spatial data collection, such as the ArcCollector App offered by ESRI, plus an external receiver for improved positional accuracy)
- Field notebooks, pencils, and data sheets (1 for each field technician collecting data)
- Copy of field protocol
- Plant field guides
- 1 1m x 1m PVC quadrat
- Digital camera
- Digital tablet (for data collection)
- Pedometer (to measure distance traveled in order to locate transects)
- Compass
- Densiometer
- Digital Calipers
- Survey pin for temporary establishment of transects
- 2.5 m length stick for sapling height measurements
- List of 4 letter codes for plant species

Vegetation Sampling Protocol

Herbivory Quantification

Moose browse will be quantified using two metrics: Accumulated Browse Index (ABI) for trees and saplings and the ratio of bites taken over bites available for saplings and shrubs. ABI is a field-based classification system that facilitates rapid assessment of moose browse intensity, and has become a well-established positive predictor of ungulate browsing (Mathisen et al. 2017). ABI will be used with sapling and tree vegetation categories, and is an architectural browse assessment tool that quantifies the amount of browsing experienced by an individual plant, based on the severity of herbivory-caused structural alteration. Plants are rated with an integer between 0 and 3, such that:

- An ABI of 0 indicates no visible evidence of moose browse.
- An ABI of 1 indicates that the plant has been previously browsed, but the structure of the tree has not changed.
- An ABI of 2 indicates that previous browsing has caused a visible change in tree structure, such as a crooked stem or increased branching.
- Finally, an ABI of 3 indicates that browsing has strongly modified the structure of the individual; such modifications can include the growth of multiple leader stems, highly stunted vertical growth, and/or the presence of brooming architecture (See Figure 9 for visual representation).

The ABI metric has the advantage of detecting past browse intensity on plants through the change in plant architecture, as past browse has been shown to have a strong positive correlation with current year browse (Mathisen et al. 2017). In general, an ABI of 3 indicates heavy browsing throughout the life of an individual plant, while an ABI of 1 or 2 may only suggest browsing from the previous year or two (and may only include browse on side shoots instead of leader shoots). Repeated browsing of saplings is common and occurs because new growth following a browse event is highly palatable and preferred forage for moose (Mathisen et al. 2017).

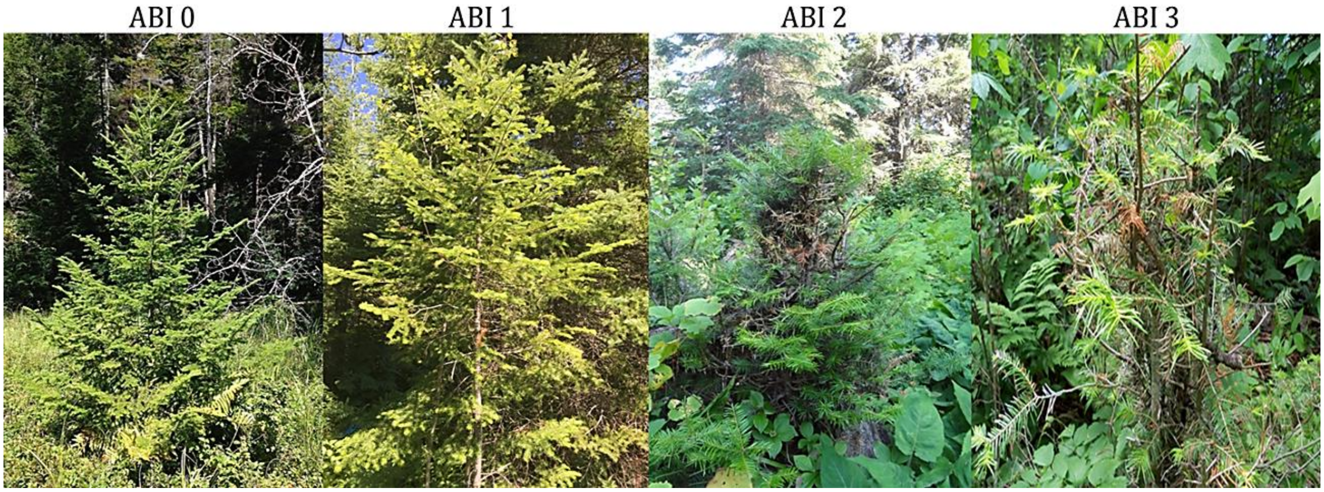


Figure 9. Examples of balsam fir individuals in each ABI class (Photo credit: J. Tourville).

In addition to ABI, moose browsing can also be assessed with biomass available/biomass removed estimates. De Jager (personal communication) has defined a standard bite size for moose (roughly 2 inches x 2 inches) (Figure 10). Using this size standard, an estimated number of individual bites removed from a plant can be obtained by recording only twigs with clear signs of browse. Similarly, any foliage remaining on a plant (the number of twigs not browsed) can be measured in the same way to generate the number of bites that are still available for moose. From these two measurements, a ratio between bites taken and bites available can be created to provide estimates of forage availability and browse intensity (when coupled with ABI measurements). This metric will be used with shrub and sapling vegetation categories.

Figure 10. Examples of typical bites taken/bites available on the foliage of a beaked hazelnut individual. Red circles indicate one bite taken each, and blue circles indicate one bite available each (Photo credit: C. Weinstein).

Herbaceous Plants and Woody Seedlings

- For the purposes of this protocol, only six indicator herbaceous species will be identified in the field. These species are highly correlated with overstory moose browse as identified by Frerker et al. (2013). These include: wild sarsaparilla (*Aralia nudicaulis*), intermediate wood fern (*Dryopteris intermedia*), western oakfern (*Gymnocarpium dryopteris*), millet grass (*Millium effusum*), mountain woodsorrel (*Oxalis montana*), and starflower (*Trientalis borealis*). All woody seedlings will also be identified.
- Indicator herbaceous plants will be classified as plants below 0.5m in height that have no persistent above-ground stems. Woody seedlings will include any predominantly woody plant species with a height less than 0.5m.
- Herbaceous plants and woody seedlings will be measured using three 1m x 1m quadrats placed at ten-meter intervals along the thirty-meter transect tape (beginning at meters 10, 20, and 30, Figure 4).

- Data recorded for herbaceous plants and woody seedlings within each 1m x 1m quadrat will include species code, species name, and common name (see sampling protocol and data sheet). Record percent cover for each species in the quadrat.
- A light availability measurement will also be taken with a denisometer centered over each quadrat.

Shrubs

- A shrub is defined as any taxa with a known shrub growth habit. A taxa needs to be primarily a multi-stemmed species with a maximum height of less than five meters. Common example genera specific for ISRO include *Alnus*, *Lonicera*, *Cornus*, and *Corylus* (see Appendix B).
- Shrubs are measured along the full length of the thirty-meter transect, and will be recorded if its foliage intersects the transect tape.
- Record species code, species name, and common name. Shrub height and the shrub's position along the transect, including the distance along the transect at which the shrub begins and ends, will be recorded. The total length of each individual shrub is the difference between these two distances.
- In order to quantify herbivory, record the number of bites taken (estimated number of bites removed from a plant by moose browse), total number of bites available (number of bites of foliage that could be available for moose browse in the future on an individual plant), and herbivory class based on the ratio of bites taken versus bites available (this later metric can be calculated after field work is complete). Herbivory class is defined as 0 (no evidence of browse), 1 (bites taken divided by bites available is ≤ .33), 2 (.33 - .66), and 3 (≥ .66). Record the diameter of three randomly selected twigs to estimate overall plant biomass.

Saplings

- Saplings are defined as any tree ≤ 2.5 meters in height, and are measured if there stems fall within two meters of the transect on either side (total survey area per transect = 120 m^2) (see Appendix A, Table 1).
- The data recorded for saplings include species code, species name, common name, as well as the individual's location left or right of transect while facing the end of the transect. Record the sapling's position in meters along the transect and its distance in meters from the transect.
- Include whether the sapling is dead or snapped. A sapling is considered snapped when an herbivore has seemingly damaged the stem, causing it to break but not sever entirely. Depending on the size of the individual, record either DBH (preferred if > 1 cm) or basal diameter.
- In order to quantify herbivory, record ABI (Accumulated Browse Index), height, number of bites taken, and the number of bites available. Record the diameter of three randomly selected browsed twigs to estimate overall plant biomass (see Discussion for rationale).

Trees

- Trees are defined as any individual >2.5 meters in height and are measured if their stems occur within two meters of the transect on either side.
- Record tree species code, species name, common name, location left or right of transect, distance from the transect in meters, distance along the transect in meters, and DBH.
- Record browse availability (1 or 0 to represent presence/absence) and ABI. Browse availability is defined as the presence of any foliage within the moose molar zone (between 0.5 meters and 2.5 meters).

Field Sampling Procedure

Transect Set-Up

1) ***ID transect :*** Record trail abbreviation, point number and transect letter (ex. if surveying the first transect of the first point on Feldtmann Lake Trail, ID would be FLT1A). The second transect (transect uptrail of the central transect when facing starting point from trail) will be called B and the central transect will be called A.
2) Place and record the first GPS point and assess forest type
3) Ensure both transects can be established and capture vegetation (not exposed rock or water). If any transect cannot be established (due to an unsafe location or water), relocate up to 200 meters away, move to other side of trail, or remove transect from sampling. Keep note of transect offset.
4) Record GPS coordinates (UTM Zone 16N, NAD83) of starting point for each transect off the trail (photos will be taken facing direction of transect in order to establish a qualitative baseline for the forest type). Measure 5 meters off trail and establish starting point for transect using survey pin. Record direction of transect using compass (cardinal direction closest to perpendicular to the trail).
5) Run transect tape 30 meters along the chosen direction selected using compass. Keep the tape close to ground level. Repeat for other transect in same cluster, except on opposite side of trail if possible.
6) If more than 2 technicians, divide into two groups: one to measure herbaceous/seedlings and shrub categories, and one to measure sapling and tree categories. In each group, one person is responsible for taking the measurements, while the other person records in either paper or digital form.
7) Record all data with either the digital tablet (with associated data collection application) or on the physical data sheets provided.

Herbaceous Plants and Woody Seedlings

1) Place 1m x 1m PVC quadrat at 10-meter intervals along transect tape (at meter 10, 20, and 30), with the target distance being the end of the quadrat (for instance, the quadrat at meter 10 will run between meter 9-10). Technicians measuring other classes must not step within one meter of

the transcect tape in these areas when sampling other plant classes if they move ahead of the herbaceous sampling team.

2) Record all herbaceous indicator and seedling species by 4 letter plant code (see Appendix A, Table 3--first two letters of genus and first two letters of species).
3) For each species, record the approximate percent cover via ocular estimation inside of quadrat.
4) Record light availability over each quadrat using the densiometer.
5) Note any other relevant observations (presence of moose pellets, hare pellets, potential rare plants, evidence of plant disease, invasive plants present).

Shrubs

1) Sampled shrubs must intersect the transect tape, be over 0.5 meters in height, and be classified as a shrub (see shrub list in Appendix A, Table 2).
2) Record species by 4 letter code.
3) Record beginning and end distance measurements for plant (in order to calculate length along transect tape).
4) Count clusters of shrubs of same species as one individual, even if it has multiple stems.
5) Measure height of each cluster at the tallest point.
6) Count number of bites removed from plant, as well as number still available (present foliage). Record as a ratio of bites taken/available.
7) Note any other observations (presence of moose pellets, hare pellets, potential rare plants, evidence of plant disease, invasive plants present).

Saplings

1) Saplings must be woody, non-shrub species between 0.5m and 2.5m in height and have stems within 2m of transect.
2) Record species by 4 letter code (Appendix A, Table 1).
3) Record if individual is on left or right side of transect tape, as well as its distance on and distance from the transect tape.
4) Record if individual is dead (i.e. no visible signs of foliage and discoloration) (dead = 1 or alive = 0), and if dead, the suspected cause of death.
5) Record DBH (at 1.37m height). If trunk is angled, record from upslope. If trunk is forked, record both main stems. Record basal DBH (at ground level) if sapling is too short for regular DBH.
6) Record ABI and bites taken/bites available as described in Herbivory Quantification section.
7) Measure height of sapling using 2.5m measuring stick.
8) Use digital calipers to measure the diameter of three browsed twigs 1 cm below the bitten end. Choose twigs at random (randomization method is choice of technician).
9) Note any other observations (presence of moose pellets, hare pellets, potential rare plants, evidence of plant disease, invasive plants present).

Trees

1) Trees must be woody, non-shrub species greater than 2.5 meters in height and have stems within 2m of transect.
2) Record all species by 4 letter code (Appendix A, Table 1).
3) Record if individual is on left or right side of transect tape, as well as its distance on and from the transect tape.
4) Record if individual is dead (0 or 1), and suspected cause of death.
5) Record DBH (at 1.37m height). If trunk is angled, record from upslope. If trunk is forked, record both main stems.
6) Record if browse is available (none available = 0 or browse available = 1) (any visible foliage below 2.5 meters in height).
7) Record ABI as described in the Herbivory Quantification section.
8) Note any other observations (presence of moose pellets, hare pellets, potential rare plants, evidence of plant disease, invasive plants present).

Data Handling

Data recorded in the field will be transcribed into a digital database immediately upon returning from the field. Long term, these data will be stored in a relational database built using Microsoft Access (provided by the authors of this report), allowing for targeted queries based on year, location, or other attributes. Any GIS data, including sampling point locations and imagery analysis outputs, will be organized within a geodatabase. All data, tabular or spatial, will follow file naming and FGDC metadata standards consistent with those currently used by NPS to ensure long-term organization and usefulness of the data. For example, following layer naming conventions set by the NPS GIS Council, a polygon layer of aquatic surface vegetation across ISRO observed in 2018 would be named similar to the following: VEG_Aquatic2018_py. At a minimum, metadata documentation needs to include purpose, method, and date that data were collected, along with descriptions of each attribute and/or data category. Possible analyses of collected data are described later in this report. We suggest that data and results be synthesized and reported on a five-year interval, depending on NPS time and resources, using a NRR data series framework.

Summary of Pilot Data

Pilot Data

The pilot data collected in the summer of 2017 followed a similar protocol as described elsewhere in this document with the exception that one transect was surveyed per point instead of the recommended two. In total, 35 individual transects were established and surveyed, spanning the far western and eastern ends of the main island in ISRO (Figure 11). Although pilot sampling consisted of one transect per point, the recommendation in this protocol is to sample two transects per point in order to more accurately capture variation in browse at a fine spatial scale. From pilot data, a summary of moose browse patterns and browse spatial trends were recorded. The example data summary below focus mainly on shrubs and saplings, as browse damage on large trees and herbaceous plants was either limited or difficult to detect (though these data would be useful in other measurements of moose browse). All summaries were conducted in R version 3.4 (R Core Development Team 2017). Data cleaning and summaries were performed using 'plyr' functions, and all graphics were generated in base R or 'ggplot2'.

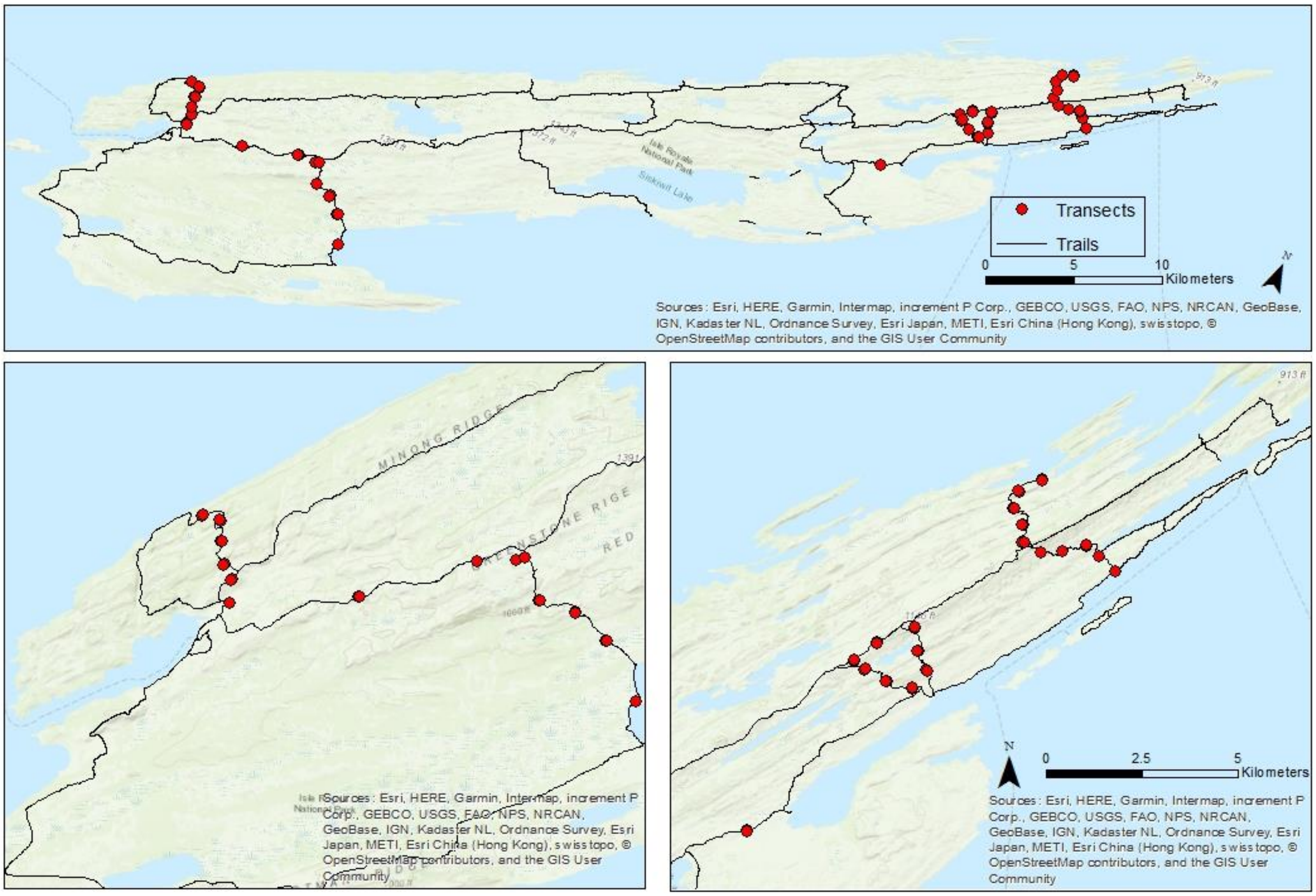


Figure 11. Map of sampled pilot transects on both the west (bottom-left) and east (bottom-right) ends of ISRO.

Species-level Browse Intensity (ABI) Measurements for Saplings

Based on data collected for the 10 most frequently sampled species: balsam fir, red maple (*Acer rubrum*), sugar maple (*Acer saccharum*), mountain maple (*Acer spicatum*), black ash (*Fraxinus nigra*), white spruce (*Picea glauca*), trembling aspen, american mountain-ash (*Sorbus americana*), showy american mountain-ash (*Sorbus decora*), and white cedar (*Thuja occidentalis*), ABI measurements suggest that trembling aspen saplings were severely browsed, with the highest proportion of level "3" ABI scores (Figure 12). In contrast, white spruce saplings showed no obvious evidence of browse, and thus received ABI scores of "0."

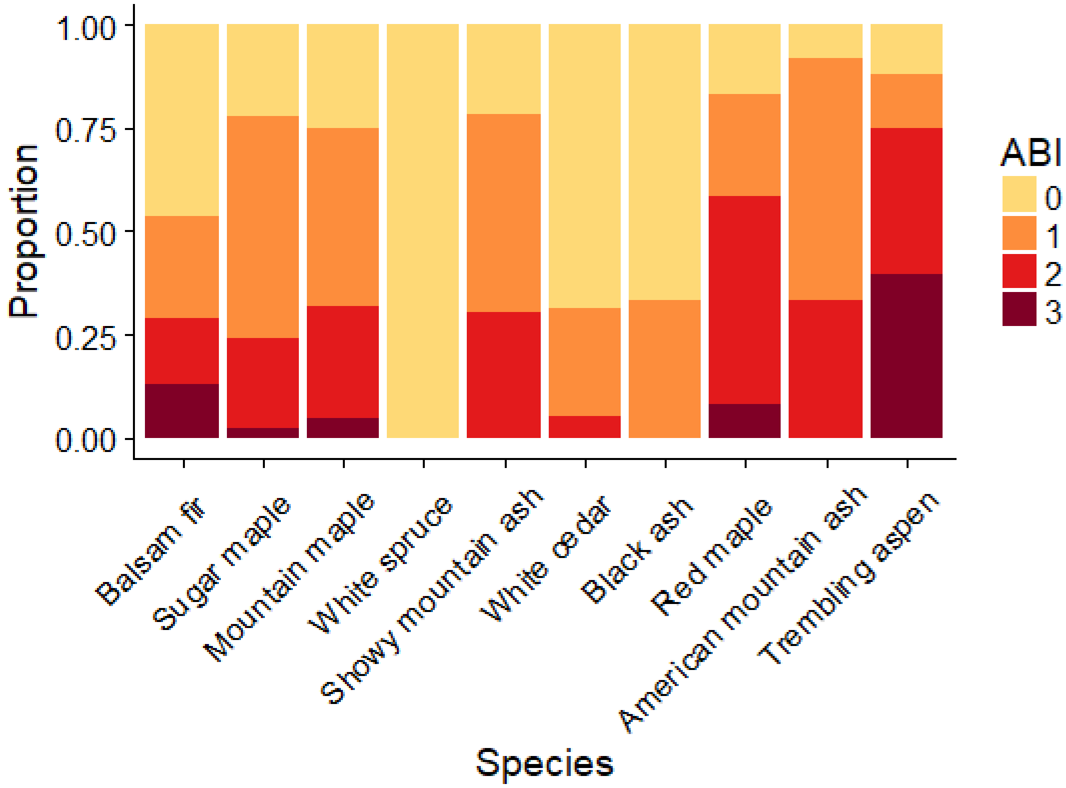


Figure 12. Browse intensity (ABI) distributions for the ten most frequently sampled sapling species.

Landscape-level ABI Measurements for Saplings

Graphs of ABI scores by species show differences among forest types (Figures 13 and 14). Moose concentrate browsing on photosynthetic and meristematic tissue, and repeated severe browsing is most hazardous to slow growing species, such as balsam fir (McLaren 1996; Pastor et al. 1998). This may in part explain the high severity of browse in balsam fir observed through pilot sampling, particularly on the Western-side of the island. Modifications to sapling architecture may have been more pronounced (high ABI scores) for balsam fir individuals due to their slower recovery rate in comparison to temperate-deciduous species, which, when seasonally available, moose are known to prefer over coniferous species (Haukioja et al. 1983; McInnes et al. 1992; Pastor et al. 1998; Peterson et al. 1998; Wam and Hjeljord 2010). In contrast, trembling aspen exhibited severe browse damage on both the Eastern- and Western-sides of the island, but available forage biomass and lost forage biomass estimates were higher on the Eastern-side of the island. This observation may reflect the rapid growth habit of trembling aspen as an early-successional species, and allude to a positive or compensational growth response to herbivory. Furthermore, Figure 15 suggests that the relative difference in “Mean Bites Taken” between an ABI value of 1 and an ABI value of 3 may be less prominent in trembling aspen, suggesting that, in proportion to biomass removed, trembling aspen has greater morphological resilience than balsam fir. However, as Mathisen et al. (2017) indicate, this resilience may trigger a “positive feedback loop,” whereby moose continue to browse on, and therefore stimulate palatable new growth.

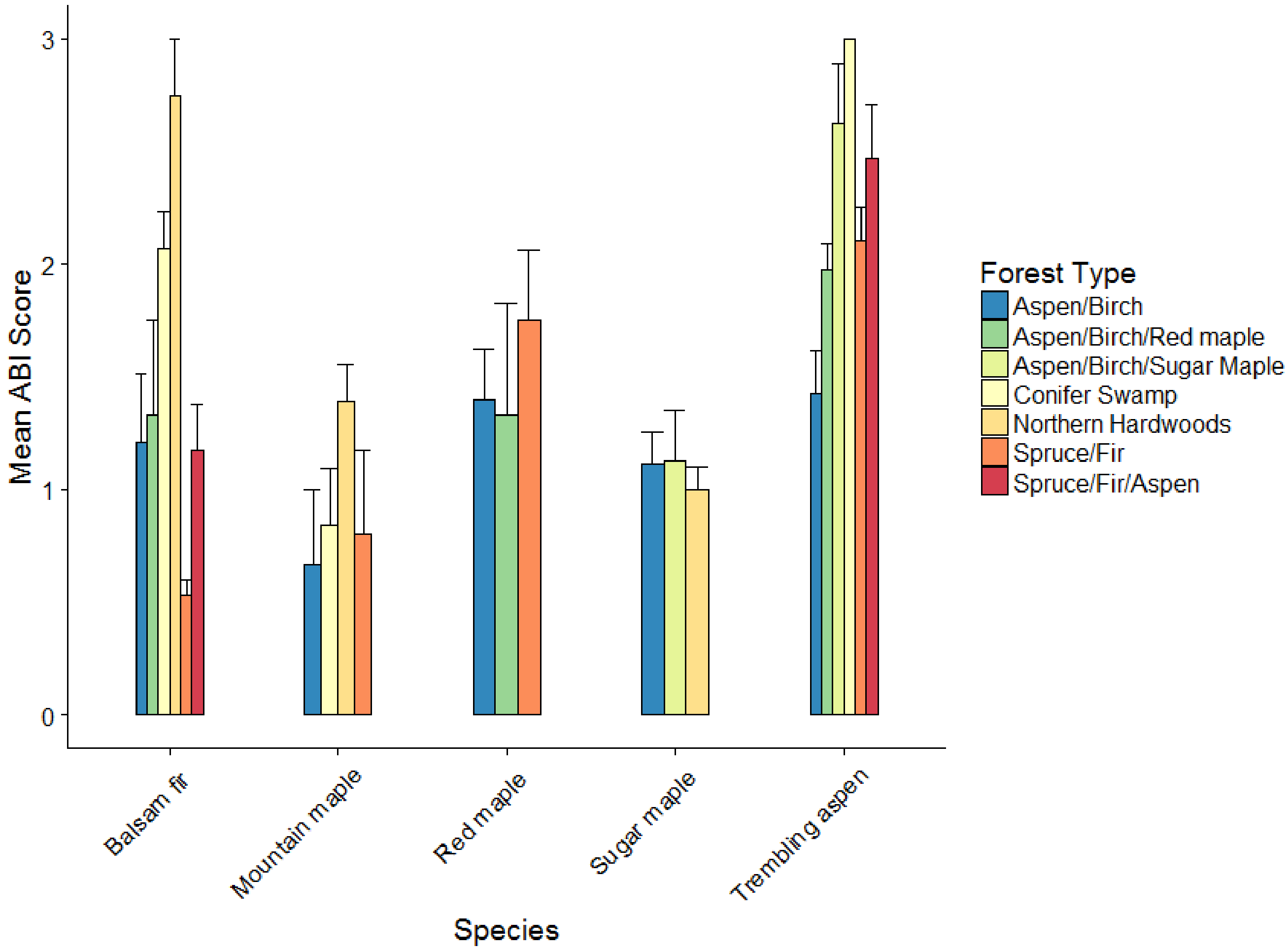


Figure 13. Sapling species mean ABI score by forest type. Error bars represent standard error.

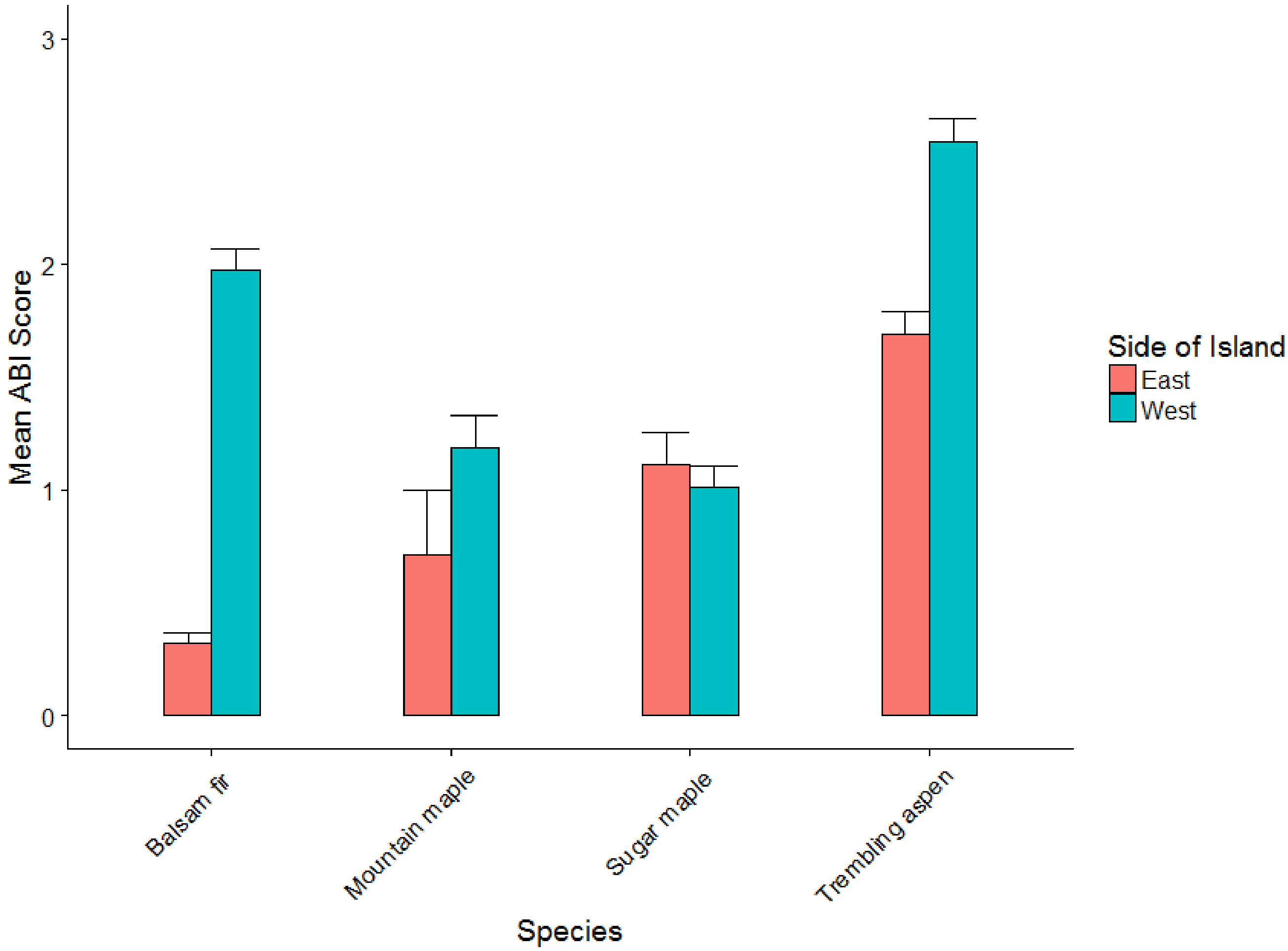


Figure 14. Sapling species mean ABI score by location (East or West-side of the island--see Figure 8 for locations for west vs. east transects). Error bars represent standard error.

Species-level Forage Estimates for Saplings

For sapling species with individuals in multiple ABI classes, the degree of biomass lost (average bites taken) generally corresponded to browse intensity, with the exception of mountain maple, in which the average number of bites taken was higher among individuals classified as ABI 3 (Figure 15). Figure 16 compares biomass removed and ABI of balsam fir and trembling aspen. This analysis demonstrates that ABI may correspond with the biomass lost (non-overlapping 95% confidence intervals), and that the relative difference in Mean Bites Taken between an ABI value of 1 and an ABI value of 3 is less pronounced for trembling aspen than for balsam fir. It may even be possible for a relationship to be established between ABI and biomass taken by moose, given more data. If possible, this may allow for quantification of biomass lost estimates directly from ABI, thereby reducing survey time in the field.

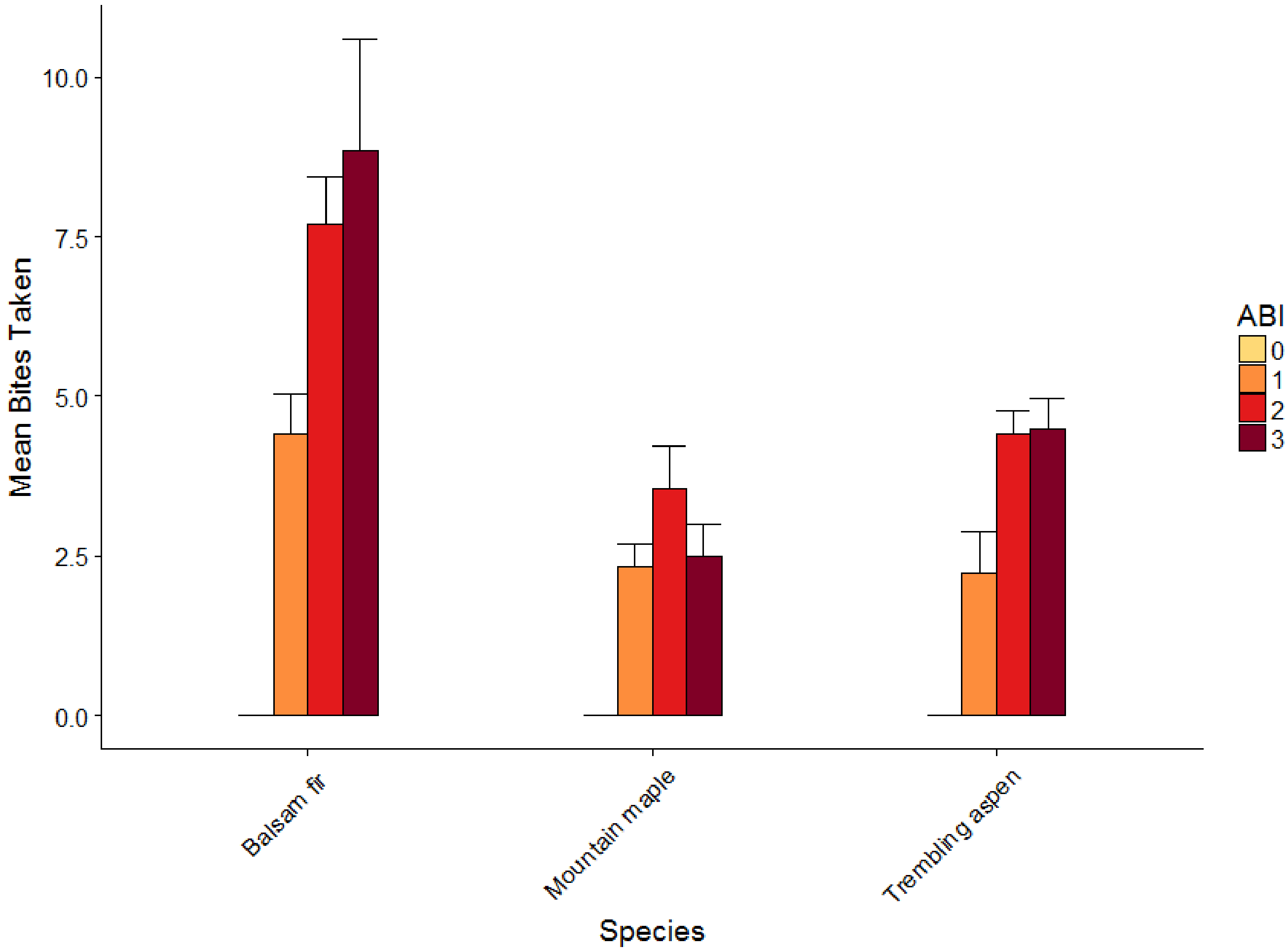


Figure 15. Biomass removed (Mean bites taken) as it relates to ABI for select sapling species. Error bars represent standard error.

Forest structure, and in particular the density of mature balsam fir and trembling aspen individuals, may also contribute to browse vulnerability (Sell 2007). As expected, the indirect relationship between tree density and ABI was more pronounced for balsam fir than for trembling aspen, as conifer architecture does not facilitate moose access and movement through the understory in the same way as the architecture of broadleaf-deciduous species. These differences in density and morphology may also help to explain the higher level of balsam fir browse on the Western-side of the island, which is dominated by northern hardwood forest, versus the Eastern-side of the island, which is dominated by boreal spruce-fir forest, as individuals on the western side are more accessible to moose (Brandner et al. 1990). Density dependence may also be influenced by environmental disturbance, such as windthrow and fire on ISRO. The most recent large fire occurred in the 1930s, and would have affected forest density in the central portion of the island (De Jager et al. 2017); however our pilot sampling points did not include this part of the island.

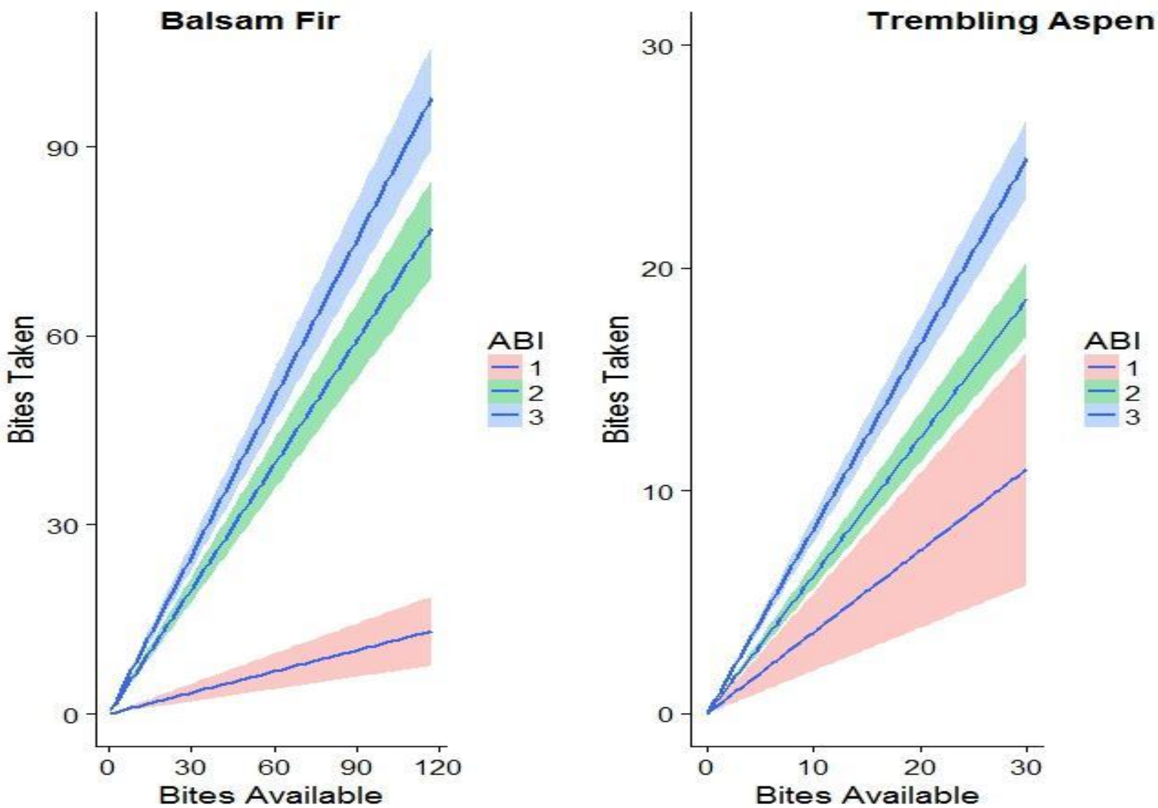


Figure 16. Relationship between biomass removed and ABI for two heavily-browsed sapling species, balsam fir and trembling aspen. Note the non-overlapping 95% confidence intervals since these indicate that plants categorized in each ABI class can predict significantly different ratios between bites available and bites taken.

Species-level Browse Intensity ("Class") Measurements for Shrubs

"Class" estimates of browse intensity for shrubs were defined as the ratio of bites taken (estimated number of bites removed from a plant by moose browse) to bites available (number of bites of foliage that could be available for moose browse in the future on an individual plant); a class of 0 indicates no evidence of browse; percentages less than or equal to 33% were grouped into class 1; percentages ranging from 33-67% were grouped into class 2; and percentages equal to or greater than 67% were considered to be class 3 (Figure 17). While the majority of shrub species showed little to no evidence of browse, beaked hazelnut (*Corylus cornuta*) had the largest proportion of browsed individuals. Although with fewer individuals sampled, serviceberries (*Amelanchier* spp.) also had a high proportion of browsed individuals. These shrub species represented a considerable portion of some transects, and thus, should be considered as a large fraction of the available forage biomass of ISRO.

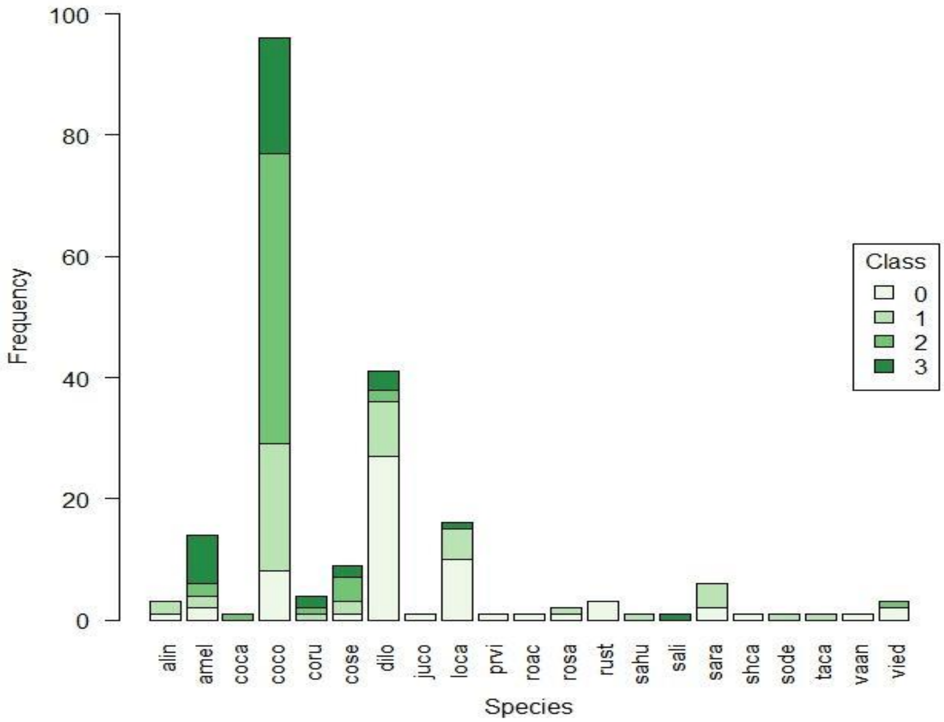


Figure 17. Browse intensity ("Class") distributions for shrub species.

In sum, data collected following this protocol will allow for the evaluation of vegetation and moose browsing changes over time and space, and constitute the basis of further modeling work aimed at predicting moose browse intensity and available forage over all of ISRO under current and forecasted climate scenarios.

Discussion

Methodological Justification

The trail-focused, transect-based design of this protocol provides the most flexible sampling scheme with the lowest amount of vegetation trampling across the landscape. Locating transects using the trail network enables technicians to quickly access each transect, capturing variation in browse intensity in a shorter time frame. Other researchers have suggested using alternative approaches to collect data on moose browse. For instance, Portinga and Moen (2015) proposed a sampling scheme using moose trails as corridors for placing sampling frames. At the patch level this approach proved to be very effective at capturing ample browse data. However, discerning moose trails was highly subjective, and identifying large browse patches added more subjectivity to site selection. Most importantly, with our approach, straight transects located on trails are less taxing for technicians to survey, and capture large amounts of data at a stand level. In addition, using a transect design allows for fewer errors in GPS location and marking. Instead of using a plot design with multiple GPS points for location (each with some inherent spatial error), the transect only requires a single point to locate. Even if the exact 30 m x 4 m location is not sampled again, the spatial and temporal distribution of the sampling effort and the thorough sampling of vegetation structure all over the island will allow for the detection of browsing and vegetation changes over time. This also alleviates the need for permanent markers, which are not in line with the management goals of a protected wilderness area. In addition, transects are effective in detecting differences of a variable along a gradient. In this case, a transect can pick up differences in moose browse given the distance from the trail. This trail effect has the potential to bias the findings of a browse study, but can be easily be accounted for if discovered. The decision to place two transects per point will allow for a comparison between coarse and fine scale browse dynamics across ISRO.

The proposed transect design serves two main functions: 1) to establish baseline metrics for vegetative communities across ISRO, and 2) to assess the spatial and temporal preference, and intensity of moose browsing across ISRO. For the first function, we choose to characterize the four previously mentioned vegetation classes: herbaceous plants/seedlings, shrubs, saplings, and adult trees.

Detecting browse on herbaceous species, especially spring ephemerals, is difficult considering that moose browsing usually removes the entire aboveground mass of the plant. By measuring the percent cover of a small group of known overstory browse indicator species, indirect methods to quantify moose browse intensity can be conducted, as has been described by Frerker et al. (2013). These particular species are positive indicators, and will increase in cover when browse is high. This will be coupled with light abundance data to better understand the factors that determine herbaceous species occurrence. Additionally, any data collected on seedlings and light availability within these quadrats can contribute to our overall understanding of tree species recruitment.

The shrub vegetation class will be monitored by intersection with the transect line. This approach has two main advantages. First, the multi-stemmed and clonal nature of shrubs makes it difficult to differentiate one individual plant from another, especially when grown in dense clusters. Treating distinct clumps along the transect line as large individuals allows for a more rapid assessment of shrub herbivory without decreasing sample size in any significant way. Second, by measuring both the length of the shrub along the transect and the mean height of the shrub, an estimate of canopy volume and hence, biomass can be calculated. For the purposes of this protocol, individual taxa will be considered shrubs based on their most common growth habit. Lastly, ABI for shrubs cannot be calculated because of the short life-span of some common shrubs, and because these species do not respond architecturally to browsing in predictable ways like other growth habits.

Sapling and tree vegetation classes are differentiated based solely on height, with the transition between sapling and tree classes set at 2.5 meters--the maximum estimated browsing height of an adult moose on ISRO (De Jager et al. 2009, 2017). ABI will be most important to measure within the sapling class, as this is the most heavily browsed plant life-stage and is altered structurally in predictable ways. Tree browse is also monitored, but it is assumed that ABI scores will be lower for this class given that trees have escaped the most negative effects of herbivory by increased height. Metrics like DBH (diameter at breast height) and height are critical for sapling biomass estimates as well. Differences between tree and sapling species' abundance, basal area, and density will elucidate the way in which forest composition is changing through time as this provides the basis for future plant demographic studies.

To provide data on herbivory, we quantified moose herbivory in two separate but interconnected ways. First, we categorized browsing by the ABI metric. Plants are ranked on an ordinal scale, 0-3. As previously discussed, an ABI of 0 indicates no previous browsing by moose, an ABI of 1 indicates that the plant has been previously browsed but the structure of the tree has not changed, an ABI of 2 indicates that previous browsing has caused a visible change in tree structure like a crooked stem or increased branching, and an ABI of 3 indicates that browsing has strongly modified the structure of the individual which can include the growth of multiple leader stems, highly stunted vertical growth, and the presence of brooming architecture (Mathisen et al. 2017). The index was originally created for browse assessment of scrub plants by large herbivores in a semi-arid savanna system in southern Africa (Skarpe et al. 2000), and adopted for use in high latitude forests in Scandinavia populated with moose herbivores. The ABI metric has the advantage of detecting past browse intensity on plants through the change in plant architecture, as past browse has been shown to be highly positively correlated with current year browse (Mathisen et al. 2017). In general, an ABI of 3 indicates heavy browsing throughout the life of an individual plant, while an ABI of 1 or 2 may only show browsing from the previous year or two (and may only include browse on side shoots instead of leader shoots). Re-browsing of saplings is common and occurs because new growth following a browse event is highly palatable, and preferred forage for moose.

Our second method for quantifying moose herbivory used biomass available/biomass removed estimates. Bites taken is essentially a proxy for plant biomass removed from a plant as a result of moose browsing, while bites available is a proxy metric for the available forage of an individual plant. The advantage of these measurements is that they can provide estimates of forage availability and browse intensity (when coupled with ABI measurements) to be used as input parameter estimates for modelling purposes. Specifically, these data can be used to better inform model outputs from the ungulate browse extension of the LANDIS-II forest succession model (Scheller et al. 2007; De Jager et al. 2017). Data from our protocol, combined with moose density and wolf activity data, will help to more accurately predict changes in forest type and available forest biomass, and thus contribute to prudent forest management on ISRO.

Recommendations: We recommend further work be dedicated to establishing a firm relationship between ABI and these biomass estimates. Furthermore, we recommend that the NPS focus their efforts on establishing the connection between bites taken/bites removed and actual biomass estimates. If actual biomass can be estimated from the bites taken/available measurements, a direct relationship between ABI and biomass can be obtained. This would allow field technicians to assess biomass removed by moose and the biomass of forage availability very rapidly, and could be used when time or park resources are limited. In order to establish this relationship, we propose two separate data collection techniques. First, when estimating the number of bites taken from a plant in the field, the diameter of three random browsed twigs or shoots should be measured (if present). This should be done on individual saplings or trees following the same methodology as Mathisen et al. (2017). Second, since the diameter at breast height and height of each sapling will be measured, the average twig diameter can be used in allometric equations to estimate both the total biomass of the plant, and the proportion of total plant foliage biomass either consumed by moose, or available for moose consumption. In order to use these species-specific allometric equations, beta coefficients for each species must be obtained. These are available from the US Forest Service, but many are old estimates (Jenkins et al. 2004). We suggest (contingent on NPS time and resources) that destructive harvesting of common tree species on ISRO be conducted in order to obtain more up to date species-specific coefficient estimates of the relationship between DBH, height, and plant biomass.

Aquatic Vegetation Monitoring

Many aquatic studies on ISRO have noted impacts of moose herbivory on aquatic vegetation (Lafrancois and Glase 2005), most recently Meeker et al. (2007). Of special consideration in this protocol is watershield (*Brasenia schreberi*). This submerged aquatic plant can be the dominant macrophyte in many ISRO lakes in terms of percent cover, and is an important food source for both waterfowl and browsing moose (Meeker et al. 2007; Kraft et al. 2010). However, Meeker et al. also describe uncertainty in the use of abundance of floating vegetation like *Nuphar variegata* or watershield as an indication of herbivory pressure, and the relationship between moose browse and aquatic macrophytes has not been explored since the late 1970s (Lafrancois and Glase 2005).

Recommendations: Due to the large number of inland lakes scattered across the island and minimal accessibility from the existing trails network, we recommend the use of remotely sensed imagery as the most efficient method to quantify change in aquatic vegetation. The Sentinel-2 Multispectral Instrument (MSI), launched and managed by the European Space Agency's Copernicus program, records imagery at 10-m spatial resolution in the visible and near-infrared wavelengths. Imagery from this satellite is openly accessible to the public for download and includes global coverage at 10-day temporal resolution.

Given the relatively coarse spatial resolution of Sentinel imagery in relation to the size of Isle Royale's inland lakes, we recommend a coarse, pixel-based calculation of Normalized Differential Vegetation Index (NDVI). Additionally, we recommend using a mask (created from the 2000 LCLU map) to limit calculations to only locations with water. Use of a mask isolates the study area to only areas known to be water cover, which reduces misleading NDVI influence from surrounding vegetation. NDVI is used as an indicator of vegetation presence and health, allowing us to infer the presence of floating water vegetation like watershield. Higher measured NDVI would imply a more robust population of watershield, and perhaps less intense moose browse, while lower NDVI values would imply the opposite. We recommend use of Google Earth Engine for this imagery analysis, due to its high computing speeds and simple data acquisition workflow.

As an alternative to NDVI calculations, heads-up digitization is another possible method for measuring abundance of floating surface vegetation like watershield. Areal calculations can then be used to assess the spread and/or contraction of floating surface vegetation, which may imply decreased/increased intensity of moose browse. This is the approach implemented by Rolf Peterson's research group for their assessment of aquatic vegetation, as noted earlier in this report (Peterson, unpublished data). One drawback of this approach is that it requires manual digitization and visual assessment of satellite imagery, which can become time-intensive, and requires a sufficiently high resolution to visually differentiate surface vegetation.

There are two key limitations to these two remote sensing approaches, the first being the relatively coarse spatial resolution of Sentinel data, compared to the resolution needed to accurately differentiate and identify floating surface vegetation. However, given the availability of spatial data at no financial cost, Sentinel is currently the best option available. The next best alternatives would be commercially available imagery, an aerial survey, or use of an unmanned aerial system. All of these alternatives should provide imagery at sufficient spatial resolution for heads-up digitization and areal statistics, though at lower feasibility.

The second limitation of these approaches is the inability to detect changes in submerged aquatic vegetation due to the physical properties of water. Water absorbs electromagnetic radiation even at shallow depths, meaning that only spectral signatures from vegetation very close to or above the lake's surface can be detected and incorporated into calculations. This limitation may be addressed by establishing a relationship between observed chlorophyll levels in the water and degree of moose browse. Further research will be required to determine the feasibility of this method.

The establishment of moose exclosures in these aquatic habitats may be useful in further specifying browse impacts on aquatic plant communities and assessing whether an increase in floating surface vegetation, which can be observed using remote sensing techniques as described above, serves as a reliable proxy for moose browse intensity.

Potential Data Analysis

Tree Density Analysis

Using "position from transect" (x) and "position on transect" (y) data collected for mature individuals of balsam fir and trembling aspen, a relationship between ABI values and tree density can be established (Appendix D, Figure 1). With our limited pilot data, it appears that areas of dense forest overstory are browsed less than more open areas. Although causality cannot be established (did moose browsing create the openings, or do they prefer to browse there, or both?), this observation does help point to areas that may be more susceptible to herbivore damage, especially if climate change leads to more frequent disturbance and forest clearing. These results also mirror the findings of Brandner et al. (1990) that found high intensity browse on balsam fir where it was less dense and surrounded by fewer conspecifics. The argument can be made here that small patches of preferred browse species are the most at risk of damage in this system. Such simple comparative analyses may help to highlight important factors that make species more vulnerable to browsing.

Browse and Forest Evaluation

The data collected from the implementation of the protocol can be used to produce summary statistics of the spatial and species-level distribution of moose browse on ISRO, similar to those above. Summarizing ABI values across species and understanding how those values change based on forest type and at a larger (island) spatial scale, exposes areas and species within the park that are most susceptible to change. Additionally, multi-scale data can be compared through time to observe any changes in these variables and note any patterns that emerge. The protocol is designed to track changes in moose browsing patterns, but also changes in forest composition and forest community migration through time. If changes in forest community ranges do occur, this should be captured by changes in the relative abundance of tree species, and their overall basal area (calculated from diameter, height, and species-specific allometric coefficients). The most effective way to visualize these changes in browse and forest composition will be through spatial modelling.

Geospatial Analyses

Long-term monitoring data could provide spatially explicit information, and could be used to predict moose browse intensity and available forage as a function of spatial variables. Additionally, climate variables in the model could be manipulated to represent potential future climate change scenarios. Using the LANDIS-II browse extension model with different climate change scenarios as described by De Jager et al. (2017) would be the best approach to evaluating the combined effects of browse and a changing climate, given the data produced through this protocol.

An example of this type of analysis is visualized below, and suggests that moose density may be negatively correlated with mean winter temperature (Appendix D, Figure 2). This could be an indirect cause, as colder temperatures define the areas of the island that boreal species predominate, and where moose are known to congregate. The potential relationship between moose density and mean winter temperature was included in the browse intensity map example (Appendix D, Figure 3). Other spatially explicit input variables (layers) included elevation, ecotype, NDVI (normalized difference vegetation index), soil type, maximum summer temperature, mean annual temperature, precipitation, and forest-type specific ABI data (calculated from pilot sampling). Spatial data layers for this analysis were obtained from the NPS Data Store and USGS Elevation Derivatives for National Applications (EDNA) database (2015); climate layers were obtained through PRISM Climate Group (2015) at Oregon State University. Spatial variable coefficients (weights) were calculated with a multiple linear regression analysis in R version 3.4 (R Core Development Team 2017). These weights were then used to build equations in ArcMap Raster Calculator (Environmental Systems Research Institute 2012) and overlay spatial and climatic data layers to calculate the browse intensity risk map (Appendix D, Figure 3). The model below only considers current climate conditions, but could be modified to accept variable parameters for altered climate scenarios.

Summary of Park-Specific Issues

Large herbivore browsing has the capacity to alter forest structure and composition (McInnes et al. 1992). A bull moose can consume 7000-9000 kg of fresh biomass per year; in order meet this high caloric demand, the composition of moose intake is a function of both availability and palatability (Persson et al. 2005; Wam and Hjeljord 2010). For example, in their study of seasonal moose-forage species composition and availability, Wam and Hjeljord (2010) found that birch (*Betula* spp.) made up 43% of all trees browsed during the summer, and 27% of all trees browsed during the winter. However, birch selection was negatively correlated with the availability of more preferred/palatable species such as rowan (*Sorbus aucuparia*), willow (*Salix* spp.), and aspen. The authors attributed this result to the fact that moose have higher caloric demand in the summer months and a slower metabolism in the winter months, such that, depending on seasonal requirements, selectivity is driven more by available forage biomass or species preference. Through quantifying moose browse with biomass available/biomass removed estimates, the application of this long-term protocol will allow the NPS to track changes in available forage and forest composition overtime.

Third-party succession effects, such as those caused by high moose population density, have the capacity to alter forest structure and shift succession of vegetation through, for example, gap creation and changes in litter composition (Grubb 1986; Rotter and Rebertus 2014). Through preferentially browsing on hardwood species, and leaving spruce species unbrowsed, moose suppress high quality litter production and trigger a pathway of succession towards a landscape with sparse canopy trees and a well-developed shrub and herbaceous understory (Pastor et al. 1987; McInnes et al. 1992; Post and Pederson 2008; Fisichelli et al. 2012). These spatial patterns that result from preferred browse are referred to as “moose-spruce savannas,” and vary from patchy canopies to extensive grasslands (Rotter and Rebertus 2014). Savannas began forming in the 1930’s during an early moose population boom, when the product of browsed hardwoods and unbrowsed spruce initiated a “disclimax;” mature spruce overstory that is not threatened or replaced results in a low density, low productivity spruce “savanna.” The ensuing decrease in forest productivity further alters soil moisture and temperature, thus initiating succession at the herbaceous level. The severity of the savanna landscape is dependent upon moose population densities and browse pressure; as such, fluctuating herbivory-vegetation dynamics create spatial and temporal landscape patterns (Pastor et al. 1998).

The slower cycling of nutrients that can result from high moose density and preferred browse alters ecosystem biomass production, regeneration, and forest composition, which all contribute to future available forage (McInnes et al. 1992; Pastor et al. 1987; Sanders and Grochowski 2013). Tracking and assessing fluctuations in the availability and corresponding use of forage biomass (functional response) overtime is thus an important metric in understanding future vegetation composition and structure, as well as ecosystem function (e.g. habitat provisioning and carbon sequestration). Research has shown that forage selectivity depends on the availability of a preferred species, as well as the proximity and accessibility of other species. Therefore, it is necessary that forage biomass availability studies encompass multiple scales, both at the species and landscape levels (The Natural Control of Animal Populations by M. E. Solomon as cited in Wam and Hjeljord 2010; Gebreyohanis 2017).

Literature Cited

Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., & Gonzalez, P. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management, 259(4), 660-684.

Bonan, G. B., & Shugart, H. H. (1989). Environmental factors and ecological processes in boreal forests. Annual review of ecology and systematics, 20(1), 1-28.

Brandner, T. A., Peterson, R. O., & Risenhoover, K. L. (1990). Balsam fir on Isle Royale: effects of moose herbivory and population density. Ecology, 71(1), 155-164.

Carlisle, D. (2000). Summary of recent analyses of Isle Royale aquatic resources. Unpublished report to Isle Royale National Park, Houghton, Michigan.

Chapin, F. S., Randerson, J. T., McGuire, A. D., Foley, J. A., & Field, C. B. (2008). Changing feedbacks in the climate–biosphere system. Frontiers in Ecology and the Environment, 6(6), 313-320.

Dale, V. H., Joyce, L. A., McNulty, S., Neilson, R. P., Ayres, M. P., Flannigan, M. D., & Simberloff, D. (2001). Climate change and forest disturbances: climate change can affect forests by altering the frequency, intensity, duration, and timing of fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, or landslides. BioScience, 51(9), 723-734.

De Jager, N. R., Pastor, J., & Hodgson, A. L. (2009). Scaling the effects of moose browsing on forage distribution, from the geometry of plant canopies to landscapes. Ecological monographs, 79(2), 281-297.

De Jager, N. R., Drohan, P. J., Miranda, B. M., Sturtevant, B. R., Stout, S. L., Royo, A. A., & Romanski, M. C. (2017). Simulating ungulate herbivory across forest landscapes: A browsing extension for LANDIS-II. Ecological Modelling, 350, 11-29.

De Jager, N.R, J.J. Rohweder, B.R. Miranda, B.R. Sturtevant, T.J. Fox, and M.C. Romanski. (2017). Modelling moose–forest interactions under different predation scenarios at Isle Royale National Park, USA. Ecological Applications 27.4: 1317-1337.

Environmental Systems Research Institute (ESRI). (2012). ArcGIS Release 10.1. Redlands, CA.

Evans, P., & Brown, C. D. (2017). The boreal–temperate forest ecotone response to climate change. Environmental Reviews, (999), 1-9.

Fisichelli, N., Frelich, L. E., & Reich, P. B. (2012). Sapling growth responses to warmer temperatures 'cooled' by browse pressure. Global Change Biology, 18(11), 3455-3463.

Fisichelli, N., Hoffman, C. H., Welling, L., Briley, L., & Rood, R. (2013). Using climate change scenarios to explore management at Isle Royale National Park: January 2013 workshop report.Natural Resource Report NPS/NRSS/CCRP/NRR—2013/714. National Park Service, Fort Collins, Colorado.

Fisichelli, N. A., Frelich, L. E., & Reich, P. B. (2014). Temperate tree expansion into adjacent boreal forest patches facilitated by warmer temperatures. Ecography, 37(2), 152-161.

Flakne, R. (2003). The Holocene vegetation history of Isle Royale National Park, Michigan, USA. Canadian Journal of Forest Research, 33(6), 1144-1166.

Frerker, K., Sonnier, G., & Waller, D. M. (2013). Browsing rates and ratios provide reliable indices of ungulate impacts on forest plant communities. Forest ecology and management, 291, 55-64.

Gebreyohanis, W. (2017). The effects of exclusion of browsing moose on above ground carbon stocks in early successional stages in boreal forests (Master's thesis, NTNU).

Goldblum, D., & Rigg, L. S. (2010). The deciduous forest–boreal forest ecotone. Geography Compass, 4(7), 701-717.

Grubb, P. J. (1986). The ecology of establishment. In Ecology and design in landscape. Edited by A.D. Bradshaw, D. A. Goode, E. H. Thorp. Blackwell Oxford pp. 83-97.

Haukioja, E., K. Kapiainen, P. Niemela, & J. Tuomi. (1983). “Plant Availability Hypothesis and Other Explanations of Herbivore Cycles: Complementary or Exclusive Alternatives?” Oikos 40(3), 419–432.

Hodgson, A. L. (2010). Temporal changes in spatial patterns of moose browse, causes and consequences (Doctoral dissertation, UNIVERSITY OF MINNESOTA).

Jager, N. R. D., Pastor, J., & Hodgson, A. L. (2009). Scaling the effects of moose browsing on forage distribution, from the geometry of plant canopies to landscapes. Ecological Monographs, 79(2), 281-297.

Janke, R. A., McKaig, D., & Raymond, R. (1978). Comparison of presettlement and modern upland boreal forests on Isle Royale National Park. Forest Science, 24(1), 115-121.

Jenkins, J. C., Chojnacky, D. C., Heath, L. S., & Birdsey, R. A. (2004). Comprehensive database of diameter-based biomass regressions for North American tree species. Gen. Tech. Rep. NE-319. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northeastern Research Station. 45 p.

Jordan, P. A., B. E. McLaren, and S. M. Sell. (2000). A summary of research of moose and related ecological topics at Isle Royale, U.S.A. Alces 36:233-267.

Kraft, G. J., Mechenich, D. J., Mechenich, C., Cook, J. E., & Seiler, S. M. (2010). Assessment of natural resource conditions: Isle Royale National Park. Natural Resource Report NPS/NRPC/WRD/NRR-2010/237.

Lafrancois, B. M. and Glase, J. (2005). Aquatic studies in National Parks of the upper Great Lakes States: past efforts and future directions. Water Resources Division Technical Report, NPS/NRWRD/NRTR-2005/334. National Park Service, Denver, Colorado.

Massé, A., & Côté, S. D. (2009). Habitat selection of a large herbivore at high density and without predation: trade-off between forage and cover? Journal of Mammalogy, 90(4), 961-970.

Mathisen, K. M., Milner, J. M., & Skarpe, C. (2017). Moose–tree interactions: rebrowsing is common across tree species. BMC Ecology, 17(1), 12.

McInnes, P. F., Naiman, R. J., Pastor, J., & Cohen, Y. (1992). Effects of moose browsing on vegetation and litter of the boreal forest, Isle Royale, Michigan, USA. Ecology, 73(6), 2059-2075.

McLaren, B.E. (1996). Plant-Specific Response to Herbivory: Simulated Browsing of Suppressed Balsam Fir on Isle Royale. Ecology 77.1: 228-235.

McLaren, B. E., & Peterson, R. O. (1994). Wolves, moose, and tree rings on Isle Royale. Science, 266(5190), 1555.

Meeker, J., A. Harris, V. Hofman, E. Judziewicz, and J. Marr. (2007). Assessment of wetland communities at Isle Royale National Park. National Park Service, Great Lakes Inventory and Monitoring Network Report GLKN/2007/04. NPS, Ashland, Wisconsin.

Morellet, N., Champely, S., Gaillard, J. M., Ballon, P., & Boscardin, Y. (2001). The browsing index: new tool uses browsing pressure to monitor deer populations. Wildlife Society Bulletin, 1243-1252.

Murie, A. (1934). The moose of Isle Royale. University of Michigan Press. 66p.

Nelson, M. P., Peterson, R. O., & Vucetich, J. A. (2008). The Isle Royale wolf–moose project: Fifty years of challenge and insight. In The George Wright Forum (Vol. 25, No. 2, pp. 98-113).

Pastor, J., & Naiman, R. J. (1992). Selective foraging and ecosystem processes in boreal forests. The American Naturalist, 139(4), 690-705.

Pastor, J., Dewey, B., Naiman, R. J., McInnes, P. F., & Cohen, Y. (1993). Moose browsing and soil fertility in the boreal forests of Isle Royale National Park. Ecology, 74(2), 467-480.

Pastor, J., R.J. Naiman, and B. Dewey. (1987). A hypothesis of the effects of moose and beaver foraging on soil nitrogen and carbon dynamics, Isle Royale. Alces 23: 107-124.

Pastor, J., Dewey, B., Moen, R., Mladenoff, D. J., White, M., & Cohen, Y. (1998). Spatial patterns in the moose–forest–soil ecosystem on Isle Royale, Michigan, USA. Ecological Applications, 8(2), 411-424.

Pastor, J., Standke, K., Farnsworth, K., Moen, R., & Cohen, Y. (1999). Further development of the Spalinger-Hobbs mechanistic foraging model for free-ranging moose. Canadian Journal of Zoology, 77(10), 1505-1512.

Persson, I. L., Danell, K., & Bergström, R. (2005). Different moose densities and accompanied changes in tree morphology and browse production. Ecological Applications, 15(4), 1296-1305.

Persson, I. L., Pastor, J., Danell, K., & Bergström, R. (2005). Impact of moose population density on the production and composition of litter in boreal forests. Oikos, 108(2), 297-306.

Peterson, R. O., Thomas, N. J., Thurber, J. M., Vucetich, J. A., & Waite, T. A. (1998). Population limitation and the wolves of Isle Royale. Journal of Mammalogy, 79(3), 828-841.

Portinga, R. L., & Moen, R. A. (2015). A novel method of performing moose browse surveys. Alces: A Journal Devoted to the Biology and Management of Moose, 51, 107-122.

Post, E., & Stenseth, N. C. (1999). Climatic variability, plant phenology, and northern ungulates. Ecology, 80(4), 1322-1339.

Post, E., & Pedersen, C. (2008). Opposing plant community responses to warming with and without herbivores. Proceedings of the National Academy of Sciences, 105(34), 12353-12358.

PRISM Climate Group, Oregon State University (2015). http://prism.oregonstate.edu, created 4 Feb 2004. United States Geological Survey. Elevation Derivatives for National Applications.

R Core Team (2013). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Risenhoover, K. L., & Maass, S. A. (1987). The influence of moose on the composition and structure of Isle Royale forests. Canadian journal of forest research, 17(5), 357-364.

Rotter, M. C., & Rebertus, A. J. (2014). Plant community development of Isle Royale's moose-spruce savannas. Botany, 93(2), 75-90.

Sanders, S., & Grochowski, J. (2013). The Forests of Isle Royale National Park: Can We Preserve This Pristine Wilderness in the Face of Climate Change? Natural Areas Journal, 33(1), 66-77.

Sanders, S., & Grochowski, J. (2014). Forest vegetation monitoring protocol version 2.0: Great lakes Inventory and Monitoring Network. National Park Service Natural resource report NPS/GLKN/Nrr—2014/799. National Park Service, Fort Collins, Colorado, USA, 1-252.

Scheller, R. M., Domingo, J. B., Sturtevant, B. R., Williams, J. S., Rudy, A., Gustafson, E. J., & Mladenoff, D. J. (2007). Design, development, and application of LANDIS-II, a spatial landscape simulation model with flexible temporal and spatial resolution. Ecological Modelling, 201(3), 409-419.

Scheller, R. M., Kretchun, A. M., Van Tuyl, S., Clark, K. L., Lucash, M. S., & Hom, J. (2012). Divergent carbon dynamics under climate change in forests with diverse soils, tree species, and land use histories. Ecosphere, 3(11), 1-16.

Schlesinger, W. H., Aneja, V. P., Chapin, F. S., Comerford, N., Gibbs, J. P., Hrabik, T., Megonigal, P., Turner, M. G., Whitaker, J. (2009). Strategic plan for scientific research in Isle Royale National Park.

Skarpe, C., Bergström, R., Bråten, A. L., & Danell, K. (2000). Browsing in a heterogeneous savanna. Ecography, 23(5), 632-640.

Sell, S.M. (2007). Interactions Between Moose and Their Primary Forage at Isle Royale National Park, Lake Superior. PhD Dissertation. University of Minnesota.

Stevens Jr, D. L., & Olsen, A. R. (2004). Spatially balanced sampling of natural resources. Journal of the American Statistical Association, 99(465), 262-278.

Tischler, K. B. (2004). Aquatic plant nutritional quality and contribution to moose diet at Isle Royale National Park (Master's Thesis, Michigan Technological University).

TNC (The Nature Conservancy). (1999). USGS-NPS Vegetation Mapping Program: classification of the vegetation of Isle Royale National Park. The Nature Conservancy. Minneapolis, Minnesota. Online. (http://biology.usgs.gov/npsveg/isro/).

USGS (United States Geological Survey). (2015). Elevation Derivatives for National Applications (EDNA).

USGS (United States Geological Survey). (2000). Isle Royale National Park spatial vegetation data. USGS, Denver, Colorado. Online. (http://biology.usgs.gov/npsveg/isro/index.html#geospatial_veg_info).

Vucetich, J. A., Peterson, R. O., & Schaefer, C. L. (2002). The effect of prey and predator densities on wolf predation. Ecology, 83(11), 3003-3013.

Vucetich, J. A., & Peterson, R. O. (2004). The influence of top–down, bottom–up and abiotic factors on the moose (Alces alces) population of Isle Royale. Proceedings of the Royal Society of London B: Biological Sciences, 271(1535), 183-189.

Vucetich, J. A., & Peterson, R. O. (2017). The population biology of Isle Royale wolves and moose: an overview. URL: www.isleroyalewolf.org

Waller, D. M., Johnson, S., Collins, R., & Williams, E. (2009). Threats posed by ungulate herbivory to forest structure and plant diversity in the upper Great Lakes region with a review of methods to assess those threats. Natural Resource Report NPS/GLKN/NRR-2009/102.

Wam, H. K., & Hjeljord, O. (2010). Moose summer and winter diets along a large scale gradient of forage availability in southern Norway. European journal of wildlife research, 56(5), 745-755.

Williams, J. W., & Jackson, S. T. (2007). Novel climates, no-analog communities, and ecological surprises. Frontiers in Ecology and the Environment, 5(9), 475-482.

Zarnetske, P. L., Skelly, D. K., & Urban, M. C. (2012). Biotic multipliers of climate change. Science, 336(6088), 1516-1518.

Appendix A

Species Lists and Codes

Table 1. ISRO tree and sapling species list

Latin name	Common name	Species symbol
Abies balsamea	Balsam fir	abba
Acer rubrum	Red maple	acru
Acer saccharum	Sugar maple	acsa
Acer spicatum	Mountain maple	acsp
Betula alleghaniensis	Yellow birch	beal
Betula papyrifera	Paper birch	bepa
Crataegus spp.	Hawthorn	crat
Fraxinus nigra	Black ash	frni
Malus pumila	Common apple	mapu
Ostrya virginiana	American hop-hornbeam	osvi
Picea glauca	White spruce	pigl
Picea mariana	Black spruce	pima
Pinus banksiana	Jack pine	piba
Pinus resinosa	Red pine	pire
Pinus strobus	White pine	pist
Populus grandidentata	Bigtooth aspen	pogr
Populus tremuloides	Trembling aspen	potr
Quercus rubra	Northern red oak	quru
Sorbus americana	American mountain ash	soam
Sorbus decora	Showy mountain ash	sode
Thuja occidentalis	White cedar	thoc

Table 2. ISRO shrub species list

Latin name	Common name	Species symbol
Alnus incana	Speckled alder	alin
Amelanchier spp.	Serviceberry	amel
Cornus canadensis	Canada dogwood	coca
Cornus rugosa	Round-leaved dogwood	coru
Cornus sericea	Red-osier dogwood	cose
Corylus cornuta	Beaked hazelnut	coco
Diervilla lonicera	Bush honeysuckle	dilo
Juniperus communis	Ground juniper	juco

Lonicera canadensis	Canada honeysuckle	loca
Prunus virginiana	Chokecherry	prvi
Rosa acicularis	Wild prickly rose	roac
Rosa spp.	Rose species	rosa
Rubus strigosus	Red raspberry	rust
Salix humilis	Upland willow	sahu
Salix sp.	Willow species	sali
Sambucus racemosa	Red elderberry	sara
Shepherdia canadensis	Soapberry	shca
Taxus canadensis	Canada yew	taca
Vaccinium angustifolium	Low sweet blueberry	vaan
Viburnum edule	Squashberry	vied

Table 3. ISRO herbaceous and seedling species list (*denotes an indicator herbaceous species).

Latin name	**Common name**	**Species code**
Abies balsamea	Balsam fir	abba
Acer rubrum	Red maple	acru
Acer saccharum	Sugar maple	acsa
Acer spicatum	Mountain maple	acsp
Amelanchier spp.	Serviceberry	amel
*Aralia nudicaulis**	Wild sarsaparilla	arnu
Betula alleghaniensis	Yellow birch	beal
Betula papyrifera	Paper birch	bepa
Cornus sericea	Red-osier dogwood	cose
Corylus cornuta	Beaked hazelnut	coco
*Dryopteris intermedia**	Intermediate woodfern	drin
Fraxinus nigra	Black ash	frni
*Gymnocarpium dryopteris**	Western oakfern	gydr
Juniperus communis	Ground juniper	juco
*Millium effusum**	Millet grass	mief
Ostrya virginiana	Hop-hornbeam	osvi
*Oxalis montana**	Mountain woodsorrel	oxmo
Picea glauca	White spruce	pigl

Picea mariana	Black spruce	pima
Pinus strobus	White pine	pist
Populus tremuloides	Trembling aspen	potr
Prunus virginiana	Chokecherry	prvi
Quercus rubra	Red oak	quru
Rubus parviflorus	Thimbleberry	rupa
Salix humilis	Upland willow	sahu
Sambucus racemosa	Red elderberry	sara
Sorbus decora	Showy mountain ash	sode
Taxus canadensis	Canada yew	taca
Thuja occidentalis	White cedar	thoc
*Trientalis borealis**	Starflower	trbo

Appendix B

Data Collection Sheet

ISRO Vegetation Sampling Data Sheet

Data Collector Initials and Date: ____________________
Trail-Point-transect ID: ____________________
Direction: ____________________
GPS Start File ID: ____________________
Picture File ID: ____________________
Canopy Type: ____________________
Notes: ____________________

Herbaceous/Seedlings

Transect	Quadrat	Spp Code	% Cover	Notes

Shrubs

Transect	Spp Code	Beg.	End	Length	Height	Bite Taken	Bite Available	Ratio	Class	Notes

Saplings

Transect	Spp Code	L or R	Pos on	Pos from	Dead?	COD	DBH	Basal DBH	ABI	Height	Bite Taken	Bite Avail	Notes/Twig Diameters

Trees

Transect	Spp Code	L or R	Pos on	Pos from	Dead?	DBH	Browse Avail?	ABI	Notes

Data Collection Metadata

Transect Information

- Trail-Point-transect ID and Date: Trail abbreviation, number of central point (starting at 1) and transect (A or B) (ex. FLT1A) and date of data collection
- Direction: 0, 90, 180, or 270 (cardinal directions)

- GPS File ID: Name of file containing GPS point/coordinates of transect starting point
- Picture File ID: Name of file containing any pictures for transect
- Canopy Type: Dominant overstory tree community

Herbaceous/seedlings

- Transect: A (central point), B (uptrail of point)
- Quadrat: 1 (10m), 2 (20m), or 3 (30m)
- Spp code: 4 letter species code (see species list)
- % Cover: Ground cover of quadrat (options are 1%, 5%, 10%, 33%, 50%, 80%, and 100%)

Shrubs

- Transect: A (central point), B (uptrail of point)
- Spp code: 4 letter species code (see species list)
- Beg: Start measurement of shrub on transect tape
- End: End measurement of shrub on transect tape
- Length: End-Beg measurements
- Height: Maximum vertical height of shrub
- Bites Taken: Number of moose bites removed from plant
- Bites Available: Number of moose bites still present on plant
- Ratio: Bites taken/Bites available
- Class: 0 (no bites taken), 1 (1-33% bites taken against bites available), 2 (33-66% bites taken against bites available), 3 (66-<100% bites taken against bites available)

Saplings

- Transect: A (central point), B (uptrail of point)
- Spp code: 4 letter species code (see species list)
- L or R: Plant on left or right side of transect tape
- Pos on: Distance along transect tape
- Pos from: Distance from transect tape
- Dead?: Dead = 1, alive = 0
- COD: Suspected cause of death (if Dead =1)
- DBH: Diameter at breast height (1.37m) of plant
- Basal DBH: Plant diameter if plant is too small for regular dbh, taken at base of plant
- ABI: Accumulated Browse Index (0, 1, 2, 3)
- Height: Maximum vertical height of sapling
- Bites Taken: Number of moose bites removed from plant
- Bites Available: Number of moose bites still present on plant
- Twig Diameters: Diameter measurements of three randomly selected browsed twigs per plant

Trees

- Transect: A (central point), B (uptrail of point)
- Spp code: 4 letter species code (see species list)
- L or R: Plant on left or right side of transect tape
- Pos on: Distance along transect tape
- Pos from: Distance from transect tape
- Dead?: Dead = 1, alive = 0
- DBH: Diameter at breast height (1.37m) of plant
- Browse Avail?: Any foliage in browse zone (0.5-2.5m)

- ABI: Accumulated Browse Index (0, 1, 2, 3)

Appendix C

Sampling Design and Schedule

Table 1. List of trail sections with specified sampling frequency (years) and intervals (km), and expected number of points and transects resulting from the sampling regime. See Appendix for suggested starting points for each trail section. “High Moose Density” and “Heterogeneous Vegetation” are presence/absence variables, with 1 denoting presence and 0 absence. A trail segment was determined to have high moose density if estimated density exceeded 3 moose/km^2. Segments intersecting with 6 or greater types of vegetation were determined to exhibit heterogeneous vegetation. “# of Points” was determined based upon the length of each trail and recommended sampling interval. “# of Transects” is calculated assuming two transects per sampling point.

Trail Name	Trail Abbreviation	Length (km)	High Moose Density	Heterogeneous Vegetation	Sampling Frequency (years)	Sampling Interval (km)	# of Points	# of Transects
Greenstone Ridge Trail South (Windigo to Ishpeming Point)	GRTS	22.81	1	0	1	2	11	22
Feldtmann Lake Trail	FLT	17.01	1	0	1	1.5	11	22
Feldtmann Ridge Trail	FRT	15.91	0	1	3	1.5	10	20
Ishpeming Trail	IT	12.27	1	1	1	2	6	12
Hatchet Lake Trail	HLT	4.36	0	1	3	1	4	8
Greenstone Ridge Trail Central (Ishpeming Point to West Chickenbone)	GRTC	17.19	0	1	3	2	8	16
Central Lakes Portage Trails	CLPT	3.17	0	1	5	0.7	4	8
East Chickenbone Loop Trail	ECLT	9.50	1	1	1	1	8	16
Minong Ridge Trail	MRT	42.16	0	1	3	3	14	28
Duncan Bay-Tobin Harbor Portage Trail	DTPT	1.08	1	1	1	0.5	2	4
Greenstone Ridge Trail North (West Chickenbone to Monument Rock)	GRTN	18.12	1	0	1	2	9	18
Tobin Harbor Trail	THT	5.00	1	0	3	1	5	10
Mt. Franklin and Lane Cove Trail	FLCT	6.87	1	1	1	0.5	13	26
Lake Richie Trail	LRT	8.43	1	1	1	1	8	16
Mt. Ojibway, Daisy Farm Trail Loop	ODFT	7.60	1	1	1	0.5	15	30
Stoll Trail	ST	5.93	1	0	3	1	5	10

Rock Harbor Trail	RHT	10.90	1	0	3	2	5	10
Lookout Louise Trail	LLT	1.50	1	0	1	0.5	3	6
Island Mine Trail	IMT	7.88	1	1	1	1	7	14
Huginnin Loop	HL	11.56	1	1	1	0.75	17	34
Indian Portage Trail	IPT	12.68	1	1	1	1	12	24
Mott Island Loop	MIL	3.73	Training					
Totals		**245.67**	**-**	**-**	-	**-**	**177**	**354**

Table 2. List of trail sections with recommended sampling schedule over the next 6 field seasons. Note that the first year of sampling, 2018, will survey all of the recommended transects (N~360). After 2018, sampling will then rotate between subsets of total recommended points based on recommended sampling frequency.

Recommended Sampling by Trail for Over the Next 6 Field Seasons											
Trail Name (Abbreviation)	**Length (km)**	**Sampling Frequency (years)**	**Sampling Interval (km)**	**# of Points**	**# of Transects**	**Total # of Points in 2018**	**Total # of Points in 2019**	**Total # of Points in 2020**	**Total # of Points in 2021**	**Total # of Points in 2022**	**Total # of Points in 2023**
Greenstone Ridge Trail South (Windigo to Ishpeming Point) (GRTS)	22.81	1	2	11	22	11	11	11	11	11	11
Feldtmann Lake Trail (FLT)	17.01	1	1.5	11	22	11	11	11	11	11	11
Feldtmann Ridge Trail (FRT)	15.91	3	1.5	10	20	10	0	0	10	0	0
Ishpeming Trail (IT)	12.27	1	2	6	12	6	6	6	6	6	6
Hatchet Lake Trail (HLT)	4.36	3	1	4	8	4	0	0	3	0	0
Greenstone Ridge Trail Central (Ishpeming Point to West Chickenbone) (GRTC)	17.19	3	2	8	16	8	0	0	8	0	0
Central Lakes Portage Trails (CLPT)	3.17	5	0.7	4	8	4	0	0	0	0	4
East Chickenbone Loop Trail (ECLT)	9.50	1	1	8	16	8	8	8	8	8	8
Minong Ridge Trail (MRT)	42.16	3	3	14	28	14	0	0	14	0	0
Duncan Bay-Tobin Harbor Portage Trail (DTPT)	1.08	1	0.5	2	4	2	2	2	2	2	2

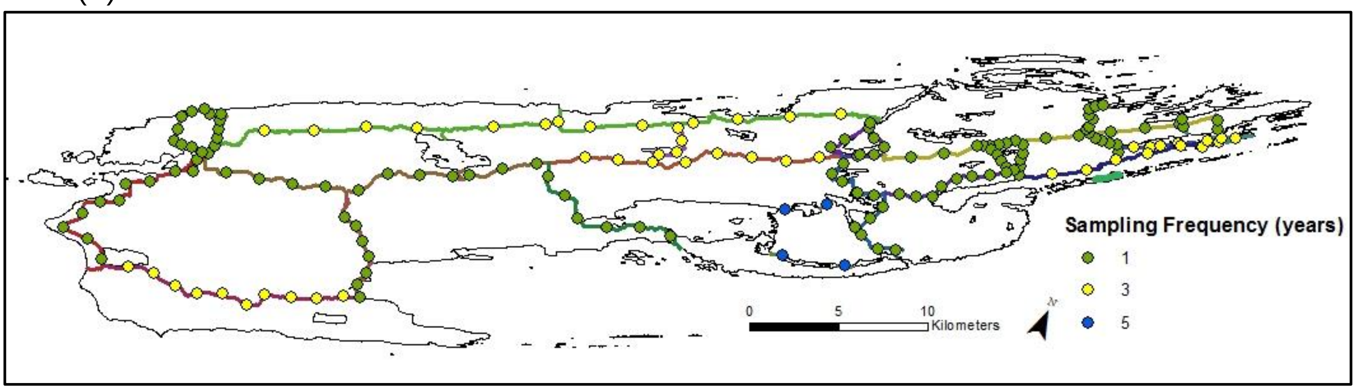
Greenstone Ridge Trail North (West Chickenbone to Monument Rock) (GRTN)	18.12	1	2	9	18	9	9	9	9	9	9
Tobin Harbor Trail (THT)	5.00	3	1	5	10	5	0	0	5	0	0
Mt. Franklin and Lane Cove Trail (FLCT)	6.87	1	0.5	13	26	13	13	13	13	13	13
Lake Richie Trail (LRT)	8.43	1	1	8	16	8	8	8	8	8	8
Mt. Ojibway, Daisy Farm Trail Loop (ODFT)	7.60	1	0.5	15	30	15	15	15	15	15	15
Stoll Trail (ST)	5.93	3	1	5	10	5	0	0	5	0	0
Rock Harbor Trail (RHT)	10.90	3	2	5	10	5	0	0	5	0	0
Lookout Louise Trail (LLT)	1.50	1	0.5	3	6	3	3	3	3	3	3
Island Mine Trail (IMT)	7.88	1	1	7	14	7	7	7	7	7	7
Huginnin Loop (HL)	11.56	1	0.75	17	34	17	17	17	17	17	17
Indian Portage Trail (IPT)	12.68	1	1	12	24	12	12	12	12	12	12
Total	**245.67**	-	-	**177**	**354**	**177**	**122**	**122**	**173**	**122**	**126**

Table 3. Suggested starting locations for trails

Trail Name	Trail Abbreviation	Starting Location
Greenstone Ridge Trail South (Windigo to Ishpeming Point)	GRTS	Windigo
Feldtmann Lake Trail	FLT	Windigo
Feldtmann Ridge Trail	FRT	From intersection with FLT
Ishpeming Trail	IT	From intersection with Greenstone
Hatchet Lake Trail	HLT	From intersection with Greenstone
Greenstone Ridge Trail Central (Ishpeming Point to West Chickenbone)	GRTC	From Ishpeming Point
Central Lakes Portage Trails	CLPT	Technician Choice
East Chickenbone Loop Trail	ECLT	From West Chickenbone
Minong Ridge Trail	MRT	From Windigo

Duncan Bay-Tobin Harbor Portage Trail	DTPT	From Moose Point
Greenstone Ridge Trail North (West Chickenbone to Monument Rock)	GRTN	From West Chickenbone
Tobin Harbor Trail	THT	From Three-mile
Mt. Franklin and Lane Cove Trail	FLCT	From Three-mile
Lake Richie Trail	LRT	From Lake Richie Camp
Mt. Ojibway, Daisy Farm Trail Loop	ODFT	From Daisy Farm
Stoll Trail	ST	From Rock Harbor
Rock Harbor Trail	RHT	From Daisy Farm
Lookout Louise Trail	LLT	From Monument Rock
Island Mine Trail	IMT	From Siskiwit Bay
Huginnin Loop	HL	From Windigo
Indian Portage Trail	IPT	From West Chickenbone

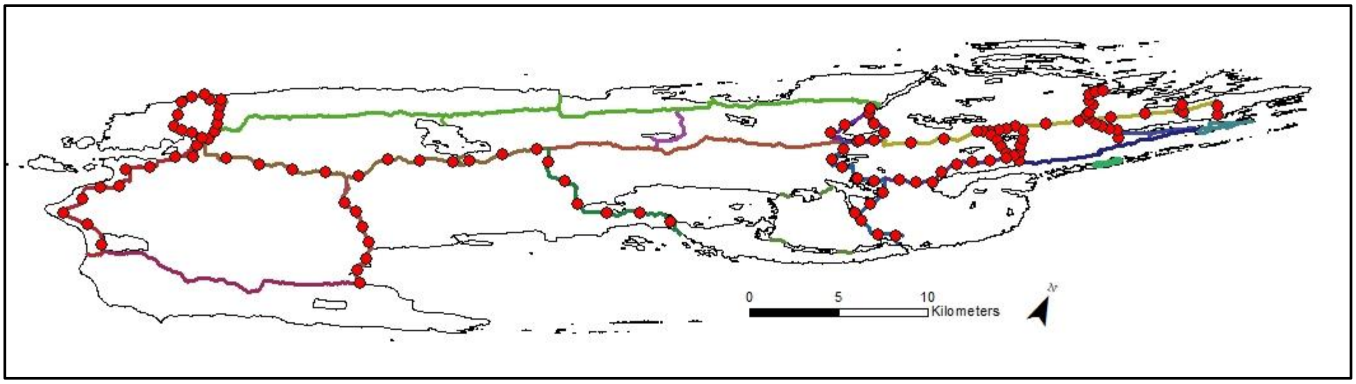
2018 (A)



Recommended Sampling by Trail for 2018 Field Season					
Trail Name (Abbreviation)	**Length (km)**	**Sampling Frequency (years)**	**Sampling Interval (km)**	**# of Points**	**# of Transects**
Greenstone Ridge Trail South (Windigo to Ishpeming Point) (GRTS)	22.81	1	2	11	22
Feldtmann Lake Trail (FLT)	17.01	1	1.5	11	22

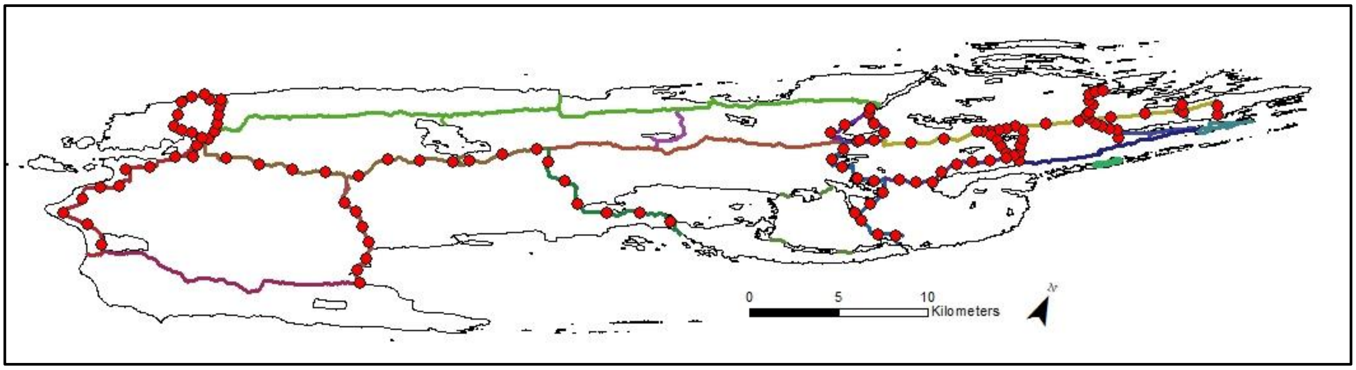
Feldtmann Ridge Trail (FRT)	15.91	3	1.5	10	20
Ishpeming Trail (IT)	12.27	1	2	6	12
Hatchet Lake Trail (HLT)	4.36	3	1	4	8
Greenstone Ridge Trail Central (Ishpeming Point to West Chickenbone) (GRTC)	17.19	3	2	8	16
Central Lakes Portage Trails (CLPT)	3.17	5	0.7	4	8
East Chickenbone Loop Trail (ECLT)	9.50	1	1	8	16
Minong Ridge Trail (MRT)	42.16	3	3	14	28
Duncan Bay-Tobin Harbor Portage Trail (DTPT)	1.08	1	0.5	2	4
Greenstone Ridge Trail North (West Chickenbone to Monument Rock) (GRTN)	18.12	1	2	9	18
Tobin Harbor Trail (THT)	5.00	3	1	5	10
Mt. Franklin and Lane Cove Trail (FLCT)	6.87	1	0.5	13	26
Lake Richie Trail (LRT)	8.43	1	1	8	16
Mt. Ojibway, Daisy Farm Trail Loop (ODFT)	7.60	1	0.5	15	30
Stoll Trail (ST)	5.93	3	1	5	10
Rock Harbor Trail (RHT)	10.90	3	2	5	10
Lookout Louise Trail (LLT)	1.50	1	0.5	3	6
Island Mine Trail (IMT)	7.88	1	1	7	14
Huginnin Loop (HL)	11.56	1	0.75	17	34
Indian Portage Trail (IPT)	12.68	1	1	12	24
Totals	**245.67**	-	**-**	**177**	**354**

2019 (B)



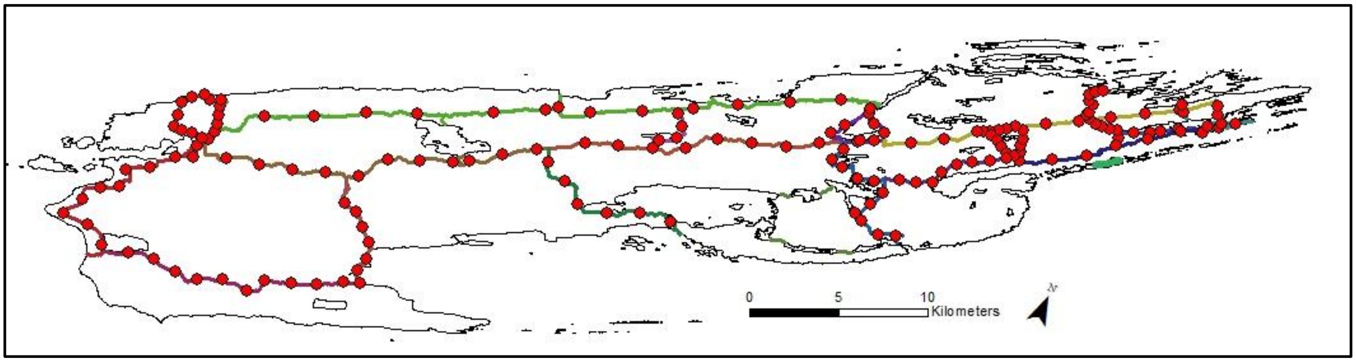
Recommended Sampling by Trail for 2019 Field Season (B)					
Trail Name (Abbreviation)	**Length (km)**	**Sampling Frequency (years)**	**Sampling Interval (km)**	**# of Points**	**# of Transects**
Greenstone Ridge Trail South (Windigo to Ishpeming Point) (GRTS)	22.81	1	2	11	22
Feldtmann Lake Trail (FLT)	17.01	1	1.5	11	22
Ishpeming Trail (IT)	12.27	1	2	6	12
East Chickenbone Loop Trail (ECLT)	9.50	1	1	8	16
Duncan Bay-Tobin Harbor Portage Trail (DTPT)	1.08	1	0.5	2	4
Greenstone Ridge Trail North (West Chickenbone to Monument Rock) (GRTN)	18.12	1	2	9	18
Mt. Franklin and Lane Cove Trail (FLCT)	6.87	1	0.5	13	26
Lake Richie Trail (LRT)	8.43	1	1	8	16
Mt. Ojibway, Daisy Farm Trail Loop (ODFT)	7.60	1	0.5	15	30
Lookout Louise Trail (LLT)	1.50	1	0.5	3	6
Island Mine Trail (IMT)	7.88	1	1	7	14
Huginnin Loop (HL)	11.56	1	0.75	17	34
Indian Portage Trail (IPT)	12.68	1	1	12	24
Totals	**137.31**	-	**-**	**122**	**244**

2020 (C)



Recommended Sampling by Trail for 2020 Field Season (C)					
Trail Name (Abbreviation)	**Length (km)**	**Sampling Frequency (years)**	**Sampling Interval (km)**	**# of Points**	**# of Transects**
Greenstone Ridge Trail South (Windigo to Ishpeming Point) (GRTS)	22.81	1	2	11	22
Feldtmann Lake Trail (FLT)	17.01	1	1.5	11	22
Ishpeming Trail (IT)	12.27	1	2	6	12
East Chickenbone Loop Trail (ECLT)	9.50	1	1	8	16
Duncan Bay-Tobin Harbor Portage Trail (DTPT)	1.08	1	0.5	2	4
Greenstone Ridge Trail North (West Chickenbone to Monument Rock) (GRTN)	18.12	1	2	9	18
Mt. Franklin and Lane Cove Trail (FLCT)	6.87	1	0.5	13	26
Lake Richie Trail (LRT)	8.43	1	1	8	16
Mt. Ojibway, Daisy Farm Trail Loop (ODFT)	7.60	1	0.5	15	30
Lookout Louise Trail (LLT)	1.50	1	0.5	3	6
Island Mine Trail (IMT)	7.88	1	1	7	14
Huginnin Loop (HL)	11.56	1	0.75	17	34
Indian Portage Trail (IPT)	12.68	1	1	12	24
Total	**137.31**	-	**-**	**122**	**244**

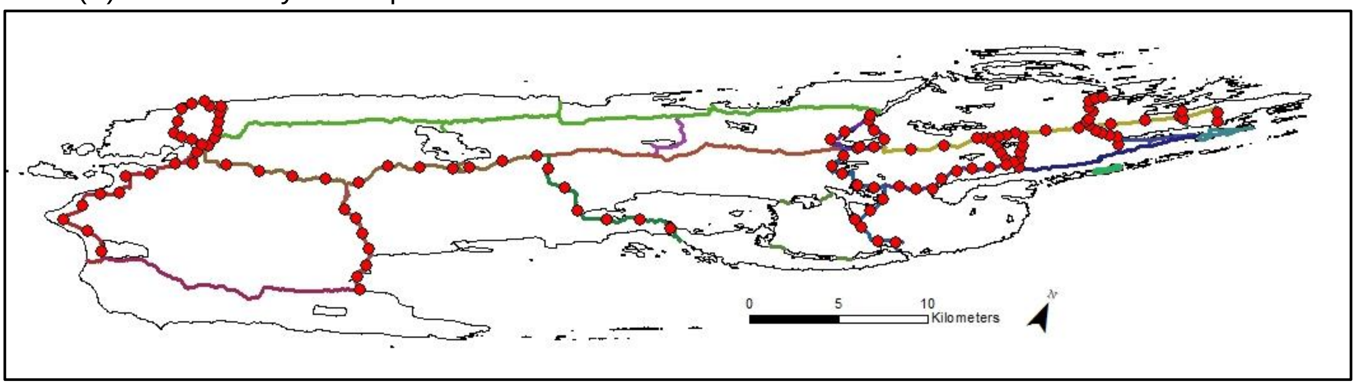
2021 (D) - Includes three year frequencies



Recommended Sampling by Trail for 2021 Field Season (D)					
Trail Name (Abbreviation)	**Length (km)**	**Sampling Frequency (years)**	**Sampling Interval (km)**	**# of Points**	**# of Transects**
Greenstone Ridge Trail South (Windigo to Ishpeming Point) (GRTS)	22.81	1	2	11	22
Feldtmann Lake Trail (FLT)	17.01	1	1.5	11	22
Feldtmann Ridge Trail (FRT)	15.91	3	1.5	10	20
Ishpeming Trail (IT)	12.27	1	2	6	12
Hatchet Lake Trail (HLT)	4.36	3	1	4	8
Greenstone Ridge Trail Central (Ishpeming Point to West Chickenbone) (GRTC)	17.19	3	2	8	16
East Chickenbone Loop Trail (ECLT)	9.50	1	1	8	16
Minong Ridge Trail (MRT)	42.16	3	3	14	28
Duncan Bay-Tobin Harbor Portage Trail (DTPT)	1.08	1	0.5	2	4
Greenstone Ridge Trail North (West Chickenbone to Monument Rock) (GRTN)	18.12	1	2	9	18
Tobin Harbor Trail (THT)	5.00	3	1	5	10
Mt. Franklin and Lane Cove Trail (FLCT)	6.87	1	0.5	13	26
Lake Richie Trail (LRT)	8.43	1	1	8	16

Mt. Ojibway, Daisy Farm Trail Loop (ODFT)	7.60	1	0.5	15	30
Stoll Trail (ST)	5.93	3	1	5	10
Rock Harbor Trail (RHT)	10.90	3	2	5	10
Lookout Louise Trail (LLT)	1.50	1	0.5	3	6
Island Mine Trail (IMT)	7.88	1	1	7	14
Huginnin Loop (HL)	11.56	1	0.75	17	34
Indian Portage Trail (IPT)	12.68	1	1	12	24
Total	**242.50**	-	**-**	**173**	**346**

2022 (E) Include five year frequencies

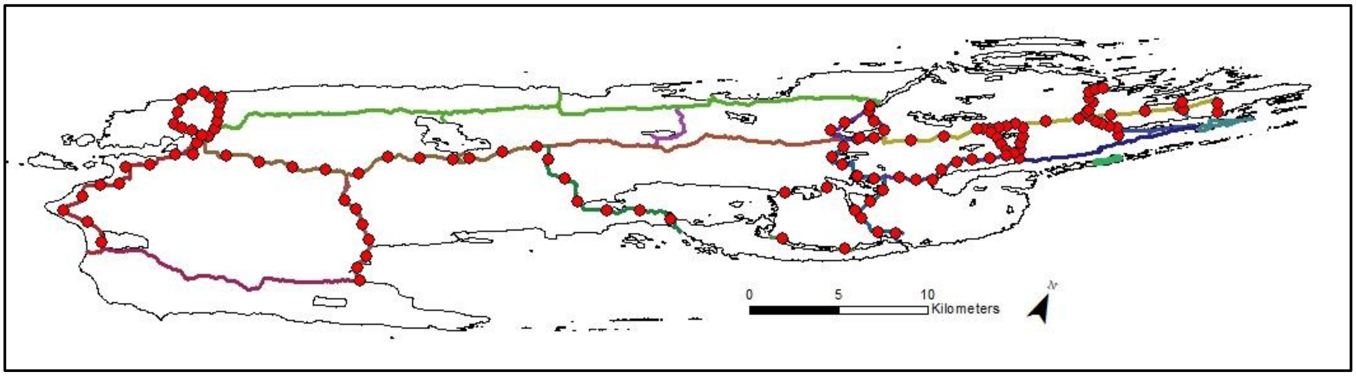


Recommended Sampling by Trail for 2023 Field Season (F)

Trail Name (Abbreviation)	Length (km)	Sampling Frequency (years)	Sampling Interval (km)	# of Points	# of Transects
Greenstone Ridge Trail South (Windigo to Ishpeming Point) (GRTS)	22.81	1	2	11	22
Feldtmann Lake Trail (FLT)	17.01	1	1.5	11	22
Ishpeming Trail (IT)	12.27	1	2	6	12
East Chickenbone Loop Trail (ECLT)	9.50	1	1	8	16
Duncan Bay-Tobin Harbor Portage Trail (DTPT)	1.08	1	0.5	2	4

Greenstone Ridge Trail North (West Chickenbone to Monument Rock) (GRTN)	18.12	1	2	9	18
Mt. Franklin and Lane Cove Trail (FLCT)	6.87	1	0.5	13	26
Lake Richie Trail (LRT)	8.43	1	1	8	16
Mt. Ojibway, Daisy Farm Trail Loop (ODFT)	7.60	1	0.5	15	30
Lookout Louise Trail (LLT)	1.50	1	0.5	3	6
Island Mine Trail (IMT)	7.88	1	1	7	14
Huginnin Loop (HL)	11.56	1	0.75	17	34
Indian Portage Trail (IPT)	12.68	1	1	12	24
Total	**137.31**	-	**-**	**122**	**244**

2023 (F) Include five year frequencies.



Recommended Sampling by Trail for 2022 Field Season (E)					
Trail Name (Abbreviation)	**Length (km)**	**Sampling Frequency (years)**	**Sampling Interval (km)**	**# of Points**	**# of Transects**
Greenstone Ridge Trail South (Windigo to Ishpeming Point) (GRTS)	22.81	1	2	11	22
Feldtmann Lake Trail (FLT)	17.01	1	1.5	11	22
Ishpeming Trail (IT)	12.27	1	2	6	12
Central Lakes Portage Trails (CLPT)	3.17	5	0.7	4	8

East Chickenbone Loop Trail (ECLT)	9.50	1	1	8	16
Duncan Bay-Tobin Harbor Portage Trail (DTPT)	1.08	1	0.5	2	4
Greenstone Ridge Trail North (West Chickenbone to Monument Rock) (GRTN)	18.12	1	2	9	18
Mt. Franklin and Lane Cove Trail (FLCT)	6.87	1	0.5	13	26
Lake Richie Trail (LRT)	8.43	1	1	8	16
Mt. Ojibway, Daisy Farm Trail Loop (ODFT)	7.60	1	0.5	15	30
Lookout Louise Trail (LLT)	1.50	1	0.5	3	6
Island Mine Trail (IMT)	7.88	1	1	7	14
Huginnin Loop (HL)	11.56	1	0.75	17	34
Indian Portage Trail (IPT)	12.68	1	1	12	24
Totals	**140.48**	-	**-**	**126**	**252**

Appendix D

Potential Data Analysis Figures

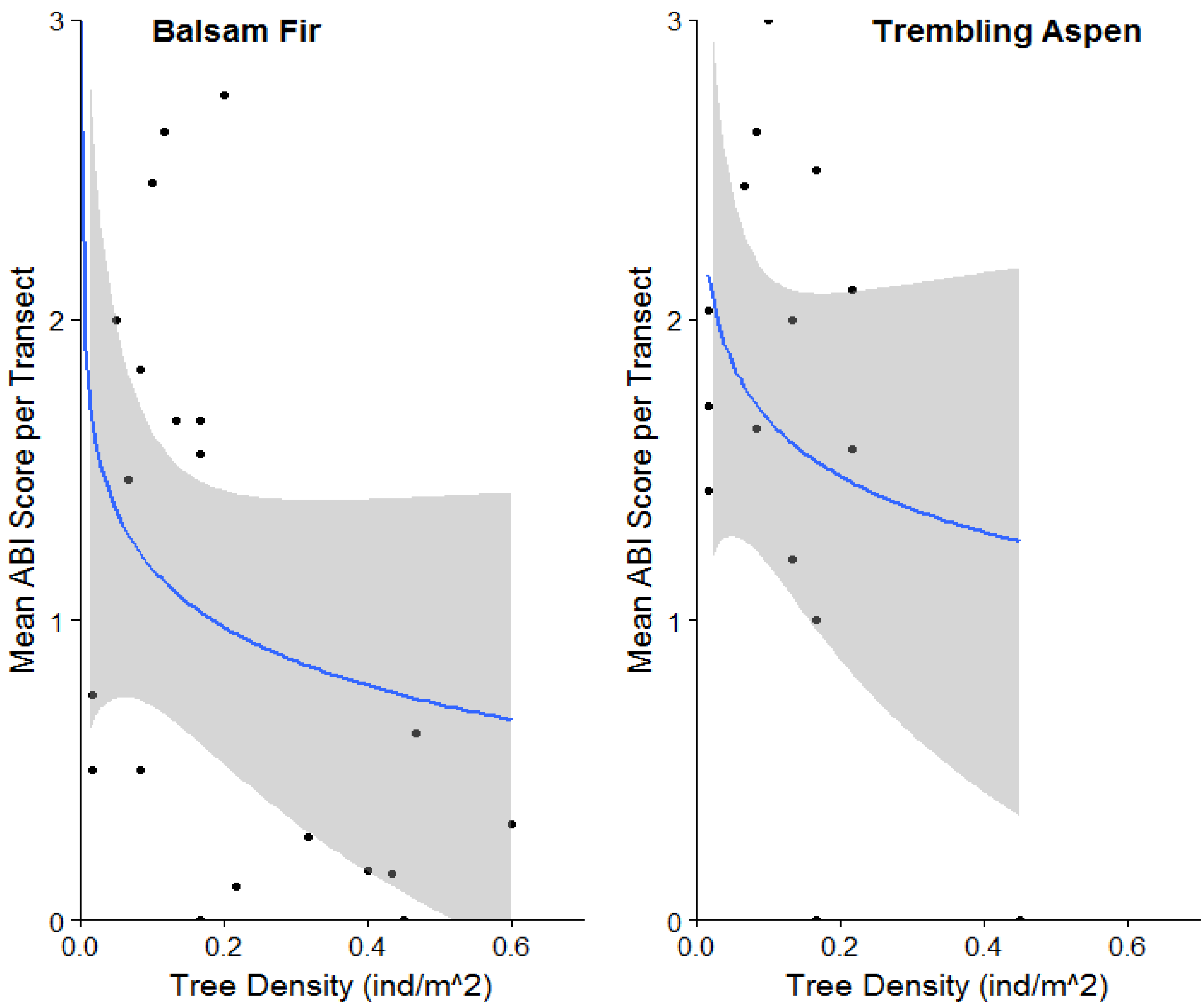


Figure 1. Mean ABI values in relation to tree density (individuals/m^2) for balsam fir and trembling aspen. Relationship follows a negative exponential decay function, and both have $R^2 > 0.35$. Shaded areas represent 95% confidence intervals.

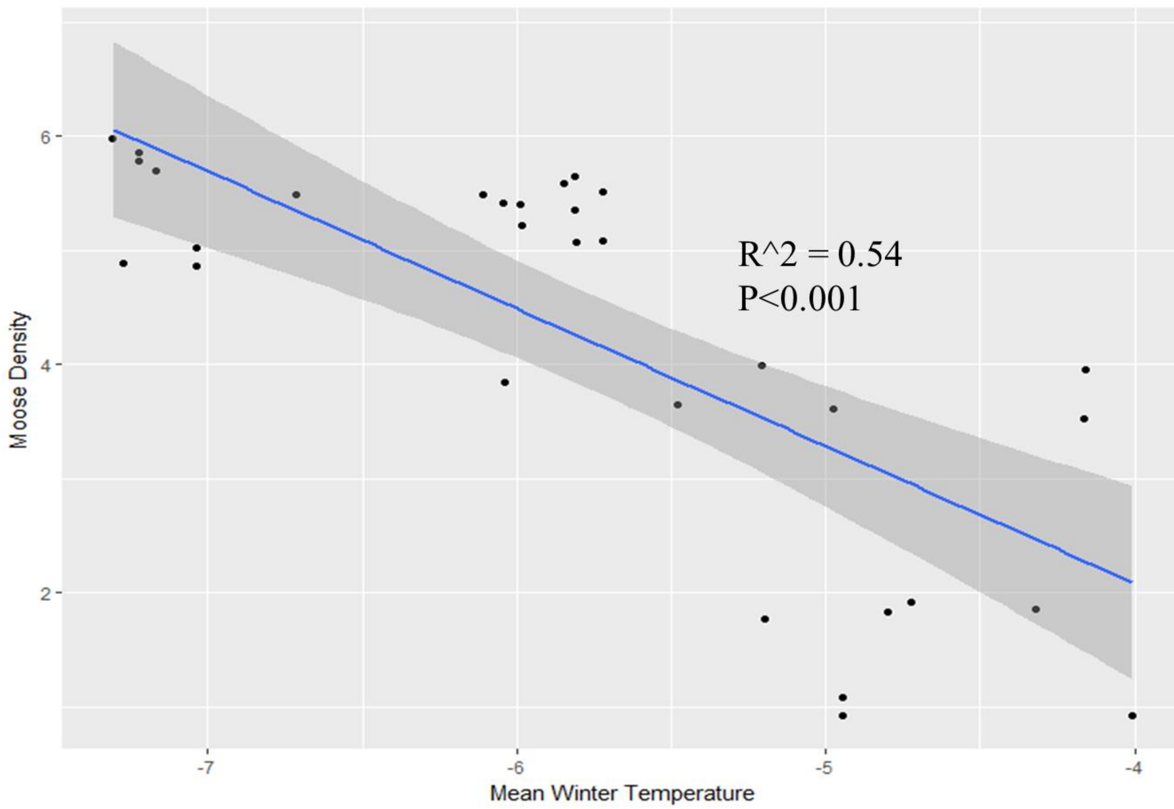


Figure 2. Relationship between mean winter temperature (in degrees Celsius) and moose density (in individuals per km^2) on ISRO. Temperature data derived from MODIS land surface temperature averaged across 2007-2017. Moose data derived from extrapolation from 2016 NPS data. Shaded area represents the 95% confidence interval.

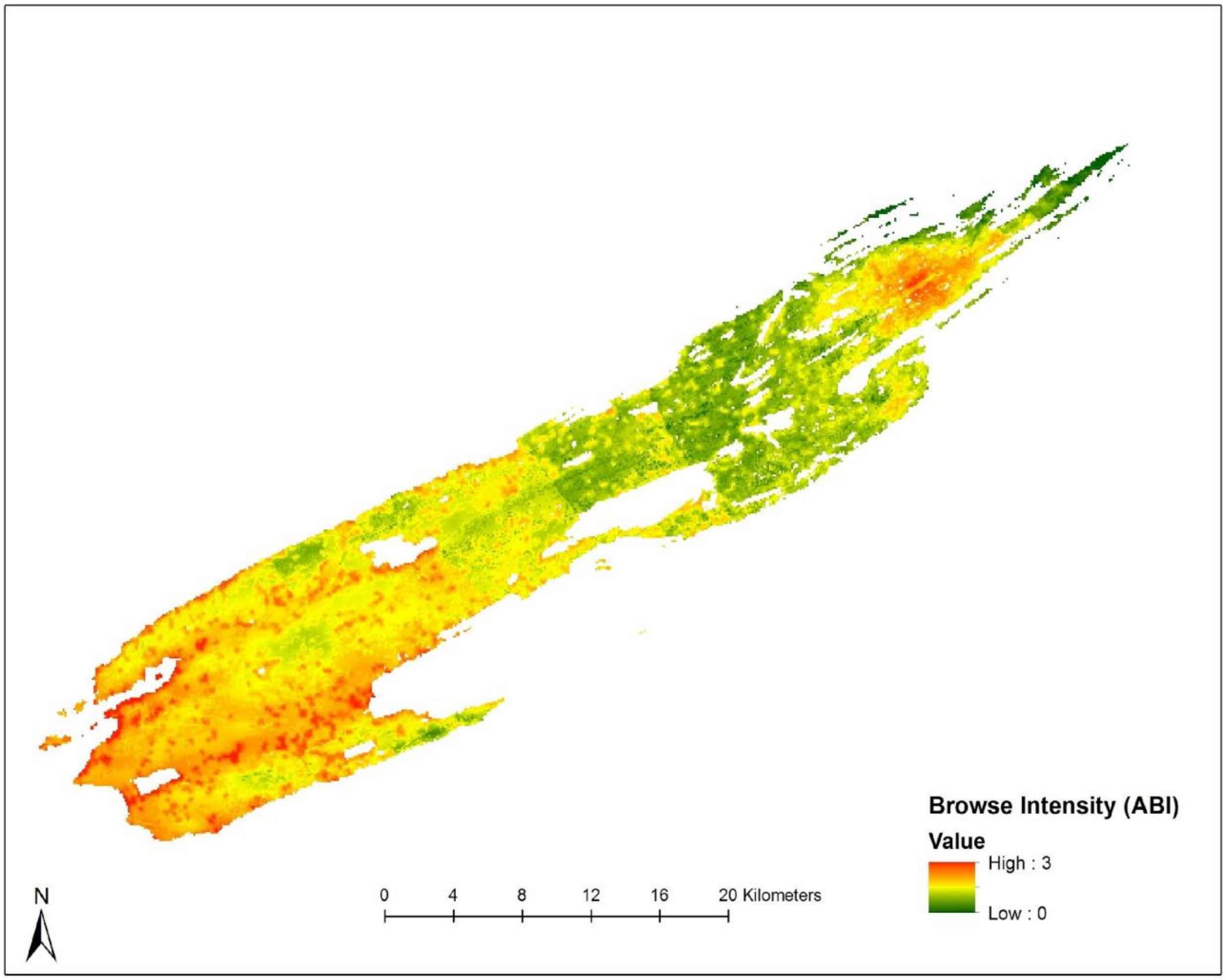


Figure 3. Browse intensity map in units of ABI for current conditions on ISRO using pilot data.

Part II: Climate Change Vulnerability Assessment of Vegetation Communities in Isle Royale National Park

National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science

Climate Change Vulnerability Assessment of Vegetation Communities in Isle Royale National Park

Natural Resource Report

ON THIS PAGE
New cones emerging on a black spruce within a northern graminoid bog
Image credit: J. Tourville

ON THE COVER
Aging spruce individuals along the rocky shore of Lake Superior
Image credit: M. Lindman and C. Weinstein

Climate Change Vulnerability Assessment of Vegetation Communities in Isle Royale National Park

Natural Resource Report

Jordon Tourville[1], Margaret Lindman[1], Charlotte Weinstein[1], Robin Schultze[1], Mirko Noack[1]

[1]University of Michigan, School for Environment and Sustainability
440 Church Street
Ann Arbor, Michigan 48109

April 2018

U.S. Department of the Interior
National Park Service
Natural Resource Stewardship and Science
Fort Collins, Colorado

Please cite this publication as:

Tourville, J. C., Lindman, M., Weinstein, C., Schultze, R., and Noack, M. 2018. Climate Change Vulnerability Assessment of Vegetation Communities in Isle Royale National Park. Natural Resource Report. National Park Service, Fort Collins, Colorado.

Contents

Figures

Page

Tables

Page

Abstract

Climate change is expected to play a major role in the restructuring of forests and other vegetation communities across the globe in the coming century. Forests in particular currently occupy almost half of the Earth's terrestrial surface, with other types of natural plant communities contributing a large proportion of land cover. Given the importance of these ecosystems with respect to their biological functions (i.e. water provisioning, climate regulation, nutrient cycling, carbon sequestration, habitat provisioning, etc.), understanding how these plant communities will respond to the growing threat of climate change will allow us to anticipate and quantify significant impacts to long-term human and environmental well-being. Climate change and subsequent rising temperatures, in concert with biotic drivers such as herbivory, is thought to have already altered the composition of plant communities in Isle Royale National Park (ISRO). Understanding what continuing changes these communities may still undergo is a major goal for National Park Service (NPS) resource managers, as this information will allow for more targeted conservation strategies. To address this need, our team has conducted a review of the available literature concerning direct and indirect climate change impacts on the vegetation communities of ISRO. Using the classification scheme of The Nature Conservancy (TNC) and the United States Geological Survey (USGS), we describe 14 unique ecological communities on ISRO, including the dominant plant species found in each community. Based on our literature review of both community- and species-level climate change impacts in this system, we present a conceptual framework for how different ecological vegetation communities and the dominant species within them may respond to the multifaceted effects of climate change on ISRO in the coming century. Our review identifies potentially "winning" and "losing" vegetative community types under commonly predicted climate change scenarios in the upper Midwest region. This synthesis of literature provides resource managers a first pass at estimating climate change effects on the plant communities of ISRO and will help in prioritizing areas for climate change mitigation.

Acknowledgments

The authors of this report would like to thank the National Park Service and specifically NPS employees Lynette Potvin and Mark Romanski for their overwhelming support and guidance for this project. We would like to also recognize Inés Ibáñez and the Global Change Ecology Lab at the University of Michigan for helpful insight with concepts and report writing. Lastly, we would like to recognize both the School for Environment and Sustainability and the University of Michigan's Matthaei Botanical Gardens and Nichols Arboretum for their generous funding, which helped support the work of this project.

List of Terms

DOC	Dissolved Oxygen Content
NPS	National Park Service
ISRO	Isle Royale National Park
TNC	The Nature Conservancy
USDA	United States Department of Agriculture
USFS	United States Forest Service
USGS	United States Geological Survey

Introduction

Statement of Task and Goals

The overall goal of this report is to synthesize the existing literature on the effects of climate change on vegetation communities, and dominant species therein, found in Isle Royale National Park (ISRO). This synthesis will be used to summarize the potential consequences of climate change (due to increasing temperature, changing precipitation regime, increased extreme event frequency, etc.) on each ecological group identified by TNC and USGS in ISRO. Through this summary, we infer which vegetation communities are likely to expand or contract across the park in the coming century due to the direct or indirect effects of a changing climate, or if individual dominant species within these communities are likely to increase or decrease in abundance. This will shed some light on how climate effects may alter plant community composition and distribution. Most importantly, our review will help to inform management action to conserve vulnerable groups.

Isle Royale National Park

Isle Royale National Park (ISRO) was established as a national park in 1940, and since then it has been only marginally impacted by anthropogenic activities (Figure 1) (Flakne 2003). These activities, performed by both Native Americans and more contemporary European settlers, included copper mining, hunting, clearing, and fires. More recently, the park has become well-known for a series of studies tracking moose herbivory and wolf predation over the past several decades (Vucetich et al. 2002; Vucetich and Peterson 2004; Nelson et al. 2008). ISRO is a federally listed National Park Service (NPS) unit composed of an archipelago of approximately 450 islands in Lake Superior, of which 99% is managed as wilderness area (Sanders and Grochowski 2013) (Figure 1). The archipelago is dominated by one main island approximately 72 km long and 14 km wide. The majority of this island is forested, with boreal spruce-fir forests dominating the northern end and shoreline, and northern hardwood forests covering the southern end (Sanders and Grochowski 2013).

Figure 1. Topographic map of ISRO (from Kraft et al. 2010)

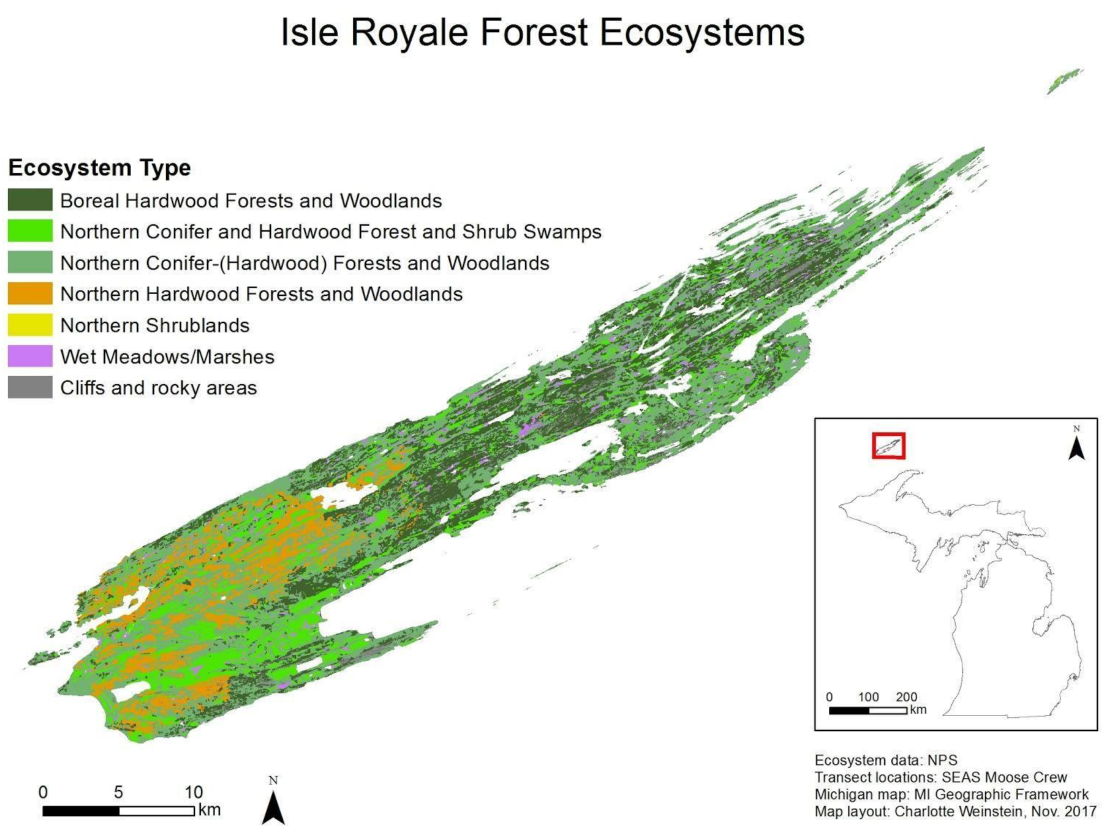


Figure 2. Map of general vegetation community types present on ISRO as delineated by the TNC and USGS (1999)

ISRO Ecological Communities

The distribution of forests and vegetation on the main island is determined largely by seasonal temperatures, soils, and topography (Figure 2) (Kraft et al. 2010). The mean annual temperature is 4.9°C, the mean January temperature is -9.7°C, and the mean July temperature is 18.8°C. The position of ISRO within a large water body like Lake Superior makes it subject to lake-effect temperature modulation, with short summers slightly cooler than the mainland, and long winters slightly warmer than the mainland. Fog and heavy precipitation are common on ISRO year-round, with a mean annual precipitation of 75 cm (Kraft et al. 2010). The areas covered by spruce-fir forests, mostly at the lowest elevations, are colder and moister on average throughout the year, with minimum winter temperatures limiting the growth of other types of forest communities in these areas (Bonan and Shugart 1989; Brandner et al. 1990; Pastor et al. 1998). The island's soils also constrain the distribution of certain forest types. Deeper and more developed alluvial soils on the southern end of the island support deciduous northern hardwood forests typical of mesic sites, while the northern end of the island is characterized by thin, dry, young, and poorly developed soils and exposed volcanic bedrock (Kraft et al. 2010). The unique geologic history of the island also plays a role in forest distribution. The island is composed of a series of parallel bedrock plates running along a southeast axis. The uplift of the island post-glaciation resulted in a topography of parallel ridges and valleys, reaching a maximum height (416 meters, minimum elevation of 179 meters) along the central ridge that runs the length of the island. The intervening valleys collect water and support black ash and alder swamps, as well as wet meadows and inland lakes (Kraft et al. 2010).

ISRO has many inland lakes, perennial and intermittent streams, and wetlands. There are 202 inland lakes across the park ranging in size from 1,635 ha (Siskiwit Lake), to numerous small and shallow ponds (Kraft et al. 2010). The lakes are a product of the unique topography of the island, coupled with glacial action. Many of the lakes are long and narrow; their shapes defined by the southwest to northeast ridge and valley orientation of the main island. The lakes have been classified based on 4 distinct categories: 1) small shallow lakes with high dissolved organic Carbon (DOC), 2) large deep lakes with low DOC, 3) lakes with hard water and high algal biomass, and 4) soft water lakes with high phosphorus (Carlisle 2000). Numerous studies cataloging aquatic taxa have been conducted in ISRO. In particular, studies highlighting aquatic macrophyte diversity (both emergent and submerged) have encountered over 400 species (Meeker et al. 2007).

Due to its isolation, Isle Royale does not support the same diversity of plant and animal species found on the closest mainland area in Minnesota (24 km to the west). However, the park does feature a number of rare plant taxa, especially on the northern end of the main island. In addition, many charismatic fauna, including moose (*Alces alces*), grey wolves (*Canis lupus*), fox (*Vulpes vulpes*), beaver (*Castor canadensis*), fish, two species of snakes, and two endangered bird species, like the peregrine falcon (*Falco peregrinus*), are known to reside in the park, and other species like the lynx and caribou have only become locally extirpated within the last century (Kraft et al. 2010).

In 1999, the Nature Conservancy (TNC), in partnership with the US Geological Survey (USGS) concluded a vegetation monitoring project across ISRO, and identified 14 distinct vegetation or forest types (ecological groups). This classification system forms the basis of our review in reference to vegetation communities. The 14 different communities include: 1) northern shrub/graminoid fens and bogs, 2) rooted/floating aquatic marshes, 3) wet meadows/marshes, 4) northern conifer and hardwood forest and shrub swamps, 5) Great Lakes rocky shores, 6) rock barrens, 7) cliffs and talus, 8) northern dry conifer (hardwood) forests and woodlands, 9) northern mesic conifer (hardwood) forests, 10) northern spruce-fir (hardwood) forests, 11) boreal hardwood forests and woodlands, 12) northern hardwood forests and woodlands, 13) northern shrublands, and 14) semi-natural meadows (TNC 1999; USGS 2000). Each of these groups will be discussed in greater detail in the results section of this report.

Compounded pressures

This review is a companion piece to our larger project, which is the creation of a long-term vegetation and herbivory monitoring protocol for ISRO. Future applications of this protocol will allow the National Park Service to track changes in moose browsing patterns over time. These data, combined with an understanding of direct and indirect climate effects on individual plant species and vegetative communities, will provide a more comprehensive picture of forest composition and forest community migration through time. Species that are both climate-vulnerable and preferred moose browse will experience compounded impacts. Assessing the combined effects of herbivory and climate change on plant species and communities will facilitate understanding of ISRO's future landscape and focus management efforts.

Climate Change

Climate Change in the Upper Midwest

The ways in which climate change impacts will manifest are expected to vary significantly by region. At a broader scale in the upper Midwest, climate change is forecasted to increase temperatures, increase rainfall, decrease snowfall, increase the length of frost-free growing season, decrease the extent of ice cover in Lake Superior, and increase the frequency of extreme storm events, causing increased frequency and severity of disturbances like fire and windthrow (Fisichelli et al. 2013; Handler et al. 2014). We consider the upper Midwest area to include the regions of Michigan, Wisconsin, and Minnesota that fall within ecoprovince 212 (Laurentian Mixed Power Forest Province) (Figure 3), as delineated by the USDA USFS (USFS 2013).

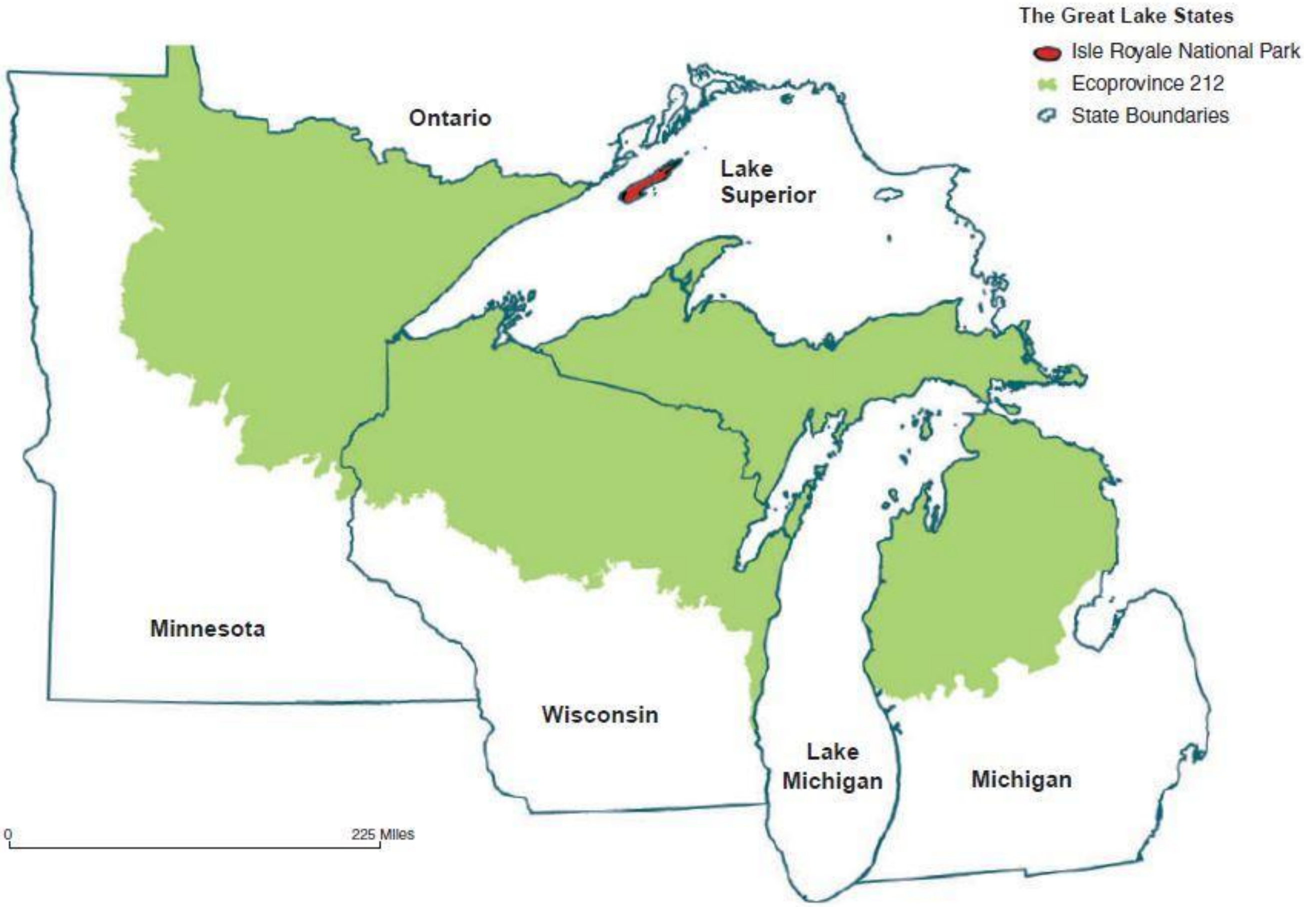


Figure 3. Extent of USFS ecoprovince 212 (from USFS 2013).

Increased temperatures contribute to higher respiration rates and more frequent drought events, both of which can impact plant communities. Both can act to decrease the carbon storage potential of plants and increase the likelihood of carbon starvation and mortality (McDowell et al. 2008; McDowell and Sevanto 2010). This may be ameliorated by the predicted increase in precipitation. However, if most yearly precipitation falls in the winter and fall months as rain instead of snow, it is possible to see very dry and drought-prone summers and wet winters (Handler et al. 2014). With an increase in temperature and less snow cover, the length of the growing season will increase. This will benefit plants in general, as a longer growing season will likely increase the amount of carbon plants fix each year. However, fast growing trees and ruderal species may benefit more from a longer growing season than shade tolerant, slow growing species, potentially altering the composition of upper Midwest forests (Dukes and Mooney 1999; Handler et al. 2014; Janowiak et al. 2018). Most importantly, the predicted frequency and severity of extreme disturbance events, such as fire and windthrow, has the potential to cause a substantial amount of damage to the structure of forests and other plant communities (Figure 4) (Dale et al. 2001). The increase in fire frequency and severity will be driven in large part by increased drought and the accumulation of subcanopy fuel. Windthrow is amplified by increasing wind storm severity and a decrease in tree health due to increased moisture stress and potential disease. These impacts may be mediated or altered by biotic interactions (interactions between different organisms), such as competition between plants and browsing on plants by herbivores (HilleRisLambers et al. 2013).

Figure 4. Examples of indirect effects of climate change in upper Midwest systems (Photo credits: J. Tourville and NPS).

Longer growing seasons may affect plant populations by increasing the resources that can be allocated to reproduction (Handler et al. 2014 and Janowiak et al. 2018). This will allow for greater individual growth and greater seed input to the community. However, this may differentially benefit some species more than others, possibly creating favorable conditions for altering community composition. Longer ice-free cover will give shoreline plant communities a relief ice scouring and damage (Dickson 2007). Microhabitat changes, as a result of lack of ice cover, for plant communities near the shore may have unexpected consequences on growth and competitive interactions between plant species. Increased fire, windthrow, and pest disturbance severity and frequency will create forest gaps, which will increase light penetration and temperature on the forest floor (Pastor et al. 1998). Fast growing species will take advantage of these conditions, again altering forest community composition, and ultimately system function (Pastor et al. 1993; Dukes and Mooney 1999).

In addition to other biotic drivers such as high herbivore pressure, the forests in the upper Midwest face several major threats as an indirect result of climate change. Exotic pests and plant species are both forecasted to increase in abundance due to warming temperatures, potentially to excluding native plants of conservation significance (Dukes et al. 2009). Spruce species are susceptible to the spruce budworm, which has the potential to eliminate large patches of standing forest and promote increased fire frequency (Dale et al. 2001). Another exotic pest currently present in this region (but not yet on ISRO) is the emerald ash borer. This insect targets all ash species, which includes the black ash commonly found in wet areas of northern Michigan (Kraft et al. 2010). In addition to exotic insect pests, invasive plants, such as invasive cattails and the common reed, have the potential to dominate inland aquatic systems, which would significantly alter the dynamics of these diverse systems (Kraft et al. 2010).

Climate change may promote the establishment and spread of invasive species in the future due to other drivers. An increase in disturbance frequency (windthrow and fire) creates conditions ideal for early successional species, like many invasives, to establish and suppress the recruitment of native plant species (Dale et al. 2001). Monitoring, prevention measures, and prompt spot treatment of exotic plant and insect outbreaks will be critical moving forward for upper Midwest resource managers.

Changes Already Noticed in ISRO

The USDA Forest Service Forest Inventory and Analysis (FIA) data has captured vegetation shifts already occurring on ISRO. Kraft et al. (2010) compare FIA data from 2008 with a thorough vegetation survey conducted in 1973 (Hansen et al. 1973). In less than 40 years, the landscape on ISRO experienced clearly detectable and quantifiable changes. Temperate (northern hardwood) forested area expanded dramatically from 10% of total forest land to nearly 23%, while boreal forest decreased from approximately 72% to 53%. Furthermore, northern white-cedar (*Thuja occidentalis*) has encroached upon much of the area formerly dominated by black spruce (*Picea mariana*). Trends in vegetation dynamics have also indicated that early successional species--such as aspen (*Populus tremuloides*) and paper birch (*Betula papyrifera*), which established around the time of European settlement--are being replaced to a greater extent by temperate species (e.g. maple (*Acer* spp.), oak (*Quercus* spp.), and white pine (*Pinus strobus*)), instead of later successional boreal coniferous species (e.g. spruce (*Picea* spp.) and fir (*Abies* spp.)). The prevailing trend of temperate forest expansion and boreal forest contraction is accentuated due to the location of ISRO on the temperate-boreal ecotone, where minor fluctuations in climate and associated effects have a heightened impact on competing forest types. These relatively recent changes in vegetation composition and distribution have occurred due to the myriad of direct and indirect consequences of climate change, such as increases in severity and frequency of disturbance (primarily windthrow), more extreme weather events, and interspecific competition between more or less resilient species, in addition to other compounding pressures, such as herbivory (Kraft et al. 2010; Fisichelli et al. 2013).

Proposed Climate Scenarios for ISRO

In their report “Using Climate Change Scenarios to Explore Management at Isle Royale National Park”, Fisichelli et al. (2013) identified and explored four plausible climate trajectories for Isle Royale through 2050 (Figure 5). The three extreme scenarios generally correspond with intensified and more frequent temperature and precipitation events and oscillations. These scenarios will most likely cause dramatic changes to the forest communities in ISRO. In all cases, boreal tree species are projected to decrease in abundance and distribution, while temperate tree species (such as sugar maple (*Acer saccharum*) and yellow birch (*Betula alleghaniensis*) are likely to increase in abundance and distribution in three of four scenarios. Thermal and water stress associated with increasing temperatures and an altered precipitation regime may limit the potential range of boreal species, which at present are strongly tied to cool and moist areas of the main island (Allen et al. 2010; Sanders and Grochowski 2013; Evans and Brown 2017). Increased interspecific competition between more thermally adapted temperate deciduous species and stressed boreal species may also play a role in the reshaping of forest communities on ISRO. Moreover, these temperate species may be able to better cope with increases in disturbance frequency than their boreal counterparts (Dale et al. 2001; Goldblum and Rigg 2010; Fisichelli et al. 2014). However, heavy browse can also limit the expansion of palatable deciduous species, potentially resulting in an increase of savanna-like systems dominated by spruce species (Rotter and Rebertus 2014).

Climate Driver	Scenario			
	Least Change	Summer Drought, Wind, and Fire	Warmer than Duluth	Isle Savanna
Mean annual temperature	**increase 3.4 °F**	same as Least Change	**increase 6.5 °F**	increase 5 °F
Cold days (< 32° F)	**15 fewer days**	same as Least Change	**up to 30 fewer days**	**up to 30 fewer days**
Hot days (> 95 °F)	increase of < 5 days	same as Least Change	increase of 5 days	same as Least Change
Growing season	**2 weeks longer**	same as Least Change	**4 weeks longer**	3 weeks longer
Annual precipitation	+5% (up Winter, down Summer)	same total as Least Change	same as Least Change	10-15% increase
Intense precipitation	20% increase in number of days with >1 inch precip	same as Least Change	same as Least Change	**summer: sporadic extreme events, 30% increase in > 1" events**
Snow	snow days -25%	same as Least Change	**snow days -50%**	**snow days -40%**
Wind	20th century conditions	**Increased probability of large wind events (derechos)**	same as Least Change	same as Least Change
Lake levels	20th century conditions	same as Least Change	same as Least Change	same as Least Change
Lake temperature	+3.6 °F in warm season temp, >50 °F water temp for 25 more days	same as Least Change	+8.3 °F in warm season temp, >50 °F water temp for 60 more days	+6.0 °F in warm season temp, >50 °F water temp for 45 more days
Lake ice cover	12 fewer days	same as Least Change	45 fewer days	30-40 fewer days
Climate variability	20th century conditions	**Punctuated dry summer periods**	**Greater variability in seasonal and annual temperature**	**Greater variability in seasonal and annual precipitation**
Arctic Oscillation	20th century conditions	same as Least Change	same as Least Change	**Predominance of positive phase (7 out of every 10 years)**

Figure 5. Detailed description of each of the climate change scenarios outlined by Fischelli et al. (2013).

Using the 14 plant communities identified by the TNC and USGS, we evaluated the response of these communities to the predicted climate change variables forecasted for the upper Midwest ecoprovince (increased temperatures, precipitation, etc.). We also evaluated the response of the 14 plant communities to the 4 climate change scenarios from the ISRO Climate Change Working Group (Fisichelli et al. 2013). In both cases, we noted if each community would likely increase or decrease in overall extent given each variable of scenario. In addition, we evaluated the response of individual dominant plant species or groups from each of the plant communities with respect to the 4 proposed climate change scenarios. Here, we present a summary of findings from the relevant literature that describes predicted changes to ISRO vegetation under forecasted climate change over the next century.

Methods

Literature relevant to this synthesis was taken from reference lists of peer-reviewed sources and government produced documents. We conducted a literature search using Web of Science and Google Scholar. Below is an example of keywords used for the northern conifer and hardwood forest and shrub swamps. Both a Web of Science and Google Scholar search was conducted for all vegetation communities. Similar syntax was used to generate a list of papers for the other vegetation communities. In addition, extensive information was extracted from a previous USFS climate change vulnerability assessment for the upper Midwest (Handler et al. 2014), and New England (Janowiak et al. 2018).

Web of Science - "TOPIC: (northern conifer swamp) OR TOPIC: (northern hardwood swamp) OR TOPIC: (shrub swamp) AND TOPIC: (climate change)"

Google Scholar - "wetland types OR classification"

Primary or secondary literature was considered relevant if it contained the specific study area (ISRO) or an analogous climate, and if it covered one or more of the following topics for each of the 14 ecological communities: description of the ecosystem under study (e.g. landform, soils, and vegetation types), characteristic species, and community-specific direct and indirect threats from climate change. Likewise, those categories were used to focus and extract information from selected studies. Date of publication was not part of the selection criteria; however the oldest study incorporated in the review is from 1973.

Literature for each community type was synthesized to assess sensitivity to changes in climatic drivers under climate change scenarios; these climate drivers include increasing temperatures, increasing or variable precipitation, decreasing snowfall, increasing length of the frost-free growing season, decreasing extent of ice cover in Lake Superior, or increasing the frequency of extreme storm events causing increased frequency and severity of disturbances, specifically fire and windthrow. This information was used to assess vegetation sensitivity to the Climate Change Working Group's proposed climate change scenarios for ISRO (Fisichelli et al. 2013). In addition, the publications that met our accepted criteria were also scanned for information on individual dominant plant species' (or major taxonomic groups) responses to climate change variables highlighted by the proposed ISRO climate change scenarios.

Results

Northern Shrub/Graminoid Fens and Bogs*

*spatial distribution not available

Northern shrubs/graminoid fens and bogs are communities that occur in layered soils. Leatherleaf bogs are the only bog type in ISRO and is an uncommon community type generally found in scattered wet depressions around the island, dominated by black spruce (*Picea mariana*) and tamarack (*Larix laricina*). Bogs are acidic and peaty, while fens on ISRO are diverse in plant composition but all alkaline. These bogs and fens typically have no drainage outlets and are waterlogged for most of the year. This community group is most commonly associated with forest types having conifer-dominated stands (TNC 1999). Species of the genus *Sphagnum* are abundant in bog and fen ecosystems and serve an important function in peat accumulation. *Sphagnum* regulates and reduces the amount of nutrients in bog and fen ecosystems by producing poor quality litter, which induces slower decomposition rates (Kool et al. 2014). *Sphagnum* also plays an important role in intercepting atmospheric nutrients, thereby restricting the number of vascular plants that can grow in this ecosystem (Kool et al. 2014).

Climate change is predicted to increase the nutrient mineralization rates in bog and fen ecosystems (Kool et al. 2014). This will increase the relative amounts of nitrogen and phosphorous available to vegetation for growth, and stimulate ecosystem productivity. Studies predict enhanced nutrient availability is expected to cause an increase in vascular plant cover while subsequently reducing the cover of *Sphagnum* (Kool et al. 2014). This shift may cause a positive feedback loop where increased vascular cover increases soil nutrient levels, which in turn facilitates further vascular plant growth and transforms the structure and composition of bog and fen ecosystems (Kool et al. 2014). In addition, *Graminoid* species are also predicted to increase at the expense of *Sphagnum* species, which may reduce the capacity of bogs and fens to form peat soils (Kool et al. 2014).

The northern shrubs/graminoid fens and bogs community group is considered to be a carbon sink because plant production exceeds the decomposition rate (Wu et al. 2014). Studies show that carbon adsorption capacity in both bog and fen ecosystems is expected to decrease as temperatures increase (Wu et al. 2014), such that these ecosystems may even switch from carbon sinks to carbon sources (Wu et al. 2014).

A further risk facing Northern shrubs/graminoid fens and bogs is the unpredictability of precipitation as climate change progresses. Evapotranspiration rates are expected to increase, potentially causing the water table levels to drop and decreasing soil moisture in these ecosystems (Wu et al. 2014). The number of heavy rain events is predicted to increase, while the intermediary conditions are drier and receive less precipitation (EFI, 2007). Water level changes are closely linked to changes in GPP (Wu et al. 2014), and water stress may constrain the optimal GPP (Wu et al. 2014).

Rooted/Floating Aquatic Marshes*

*spatial distribution not available

Rooted/floating aquatic marshes are ecological ecosystems where herbaceous plant communities float in shallow lakes and streams. The soils are characterized by loose, poorly consolidated organic matter, which developed from submerged plant beds (Kost et al. 2007). The dominant vegetation includes non-rooted submergent plants, rooted floating-leaved plants, and non-rooted floating plants (Kost et al. 2007). Common species include waterweed (*Elodea* spp.), water star-grass (*Heteranthera dubia*), milfoils (*Myriophyllum* spp.), pondweeds (*Potamogeton* spp.), water crowfoots (*Ranunculus* spp.), water-celery (*Apium graveolens*), and stoneworts (*Chara* spp.) (Kost et al. 2007).

These ecosystems emerge in areas where precipitation exceeds evapotranspiration (Short et al. 2016), creating a saturated water table for most of the year. Climate change impacts--including increased temperature, altered precipitation rates, and decline in Lake Superior water levels (Sea Grant 2005)--are particularly threatening to rooted/floating aquatic marshes, as they may disrupt the hydrological regime, which is necessary for the productivity of this ecosystem (Short et al. 2016).

Higher temperatures can lead to increased metabolic rates for species living in this ecosystem. Increased metabolism accelerates water loss, leading to lower water tables (Short et al. 2016). Studies have shown that higher temperatures led to a migration of this ecosystem towards wetter sites (Short et al. 2016). Precipitation rates are expected to increasingly fluctuate, which can lead to lower water levels and shorter periods of inundation (Short et al. 2016). This may cause a shift in species composition from hydric species to xeric species (Short et al. 2016) and devastate rooted/floating aquatic marshes.

Wet Meadows/Marshes

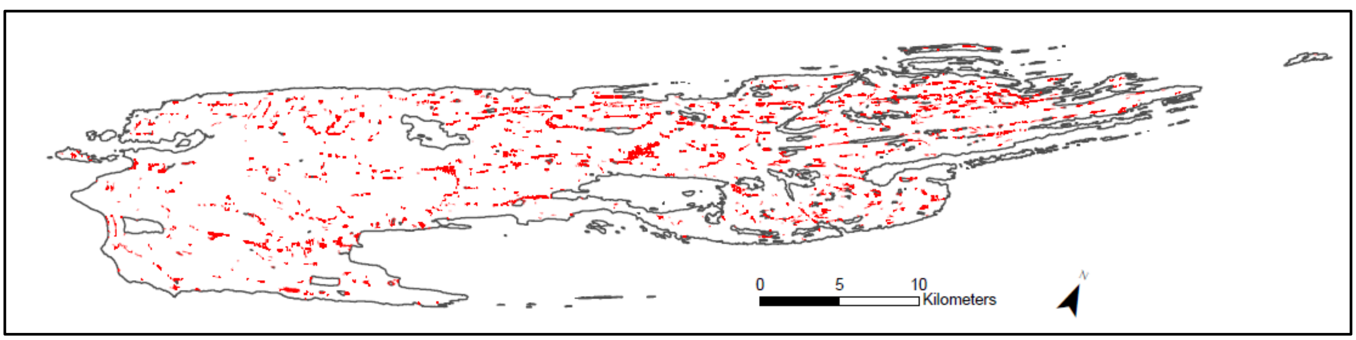


Figure 6. Distribution of the wet meadows/marshes ecological group on ISRO (4.00% of mapped vegetation cover). Map created using TNC GIS data (1999).

Wet meadows/marshes are ecological ecosystems that occur in poorly drained areas, such as shallow lake basins or between shallow marshes and upland areas. This ecosystem is usually drier than other marshes except during periods of seasonal high precipitation. Wet meadows usually lack standing water for most of the year; however, the high water table typically facilitates saturated soil throughout the year. The soils are composed of highly fertile organic matter, such as well-decomposed peat (Kost et al.

2007). The vegetation that dominate this ecosystem is constituted by grass (e.g. bluejoint grass (*Calamagrostis canadensis*)) and sedge species (e.g. tussock sedge (*Carex stricta*)) (Kost et al. 2007).

Wet meadows/marshes are highly dependent on seasonal rainfall, particularly in spring. Their primary threat due to climate change is the quality and quantity of water supply. Droughts are expected to increase due to increases in temperatures without an increases in precipitation (EFI 2007). While overall precipitation rates are predicted to increase, the number of rain days is predicted to decrease, as heavy rain events become more frequent with drier intermediary conditions (EFI 2007).

Additionally, higher temperatures can lead to increased metabolic rates for wet meadow/marsh species. Increased metabolism accelerates water loss leading to lower water levels (Short et al. 2016). Climate change is predicted to decrease the amount of freeze free days, thereby causing earlier onset of photosynthesis in spring and prolonged photosynthetic activity in autumn (EFI 2007). Earlier and prolonged photosynthetic activity coupled with potentially higher metabolic rates can increase water stress in wet meadows/marshes (EFI 2007), causing a shift in species composition from hydric species to xeric species (Short et al. 2016). Because wet meadows/ marshes are usually located in areas between shallow marshes and upland areas, species will likely migrate towards lower, wetter shallow marshes.

Northern Conifer and Hardwood Forest and Shrub Swamps

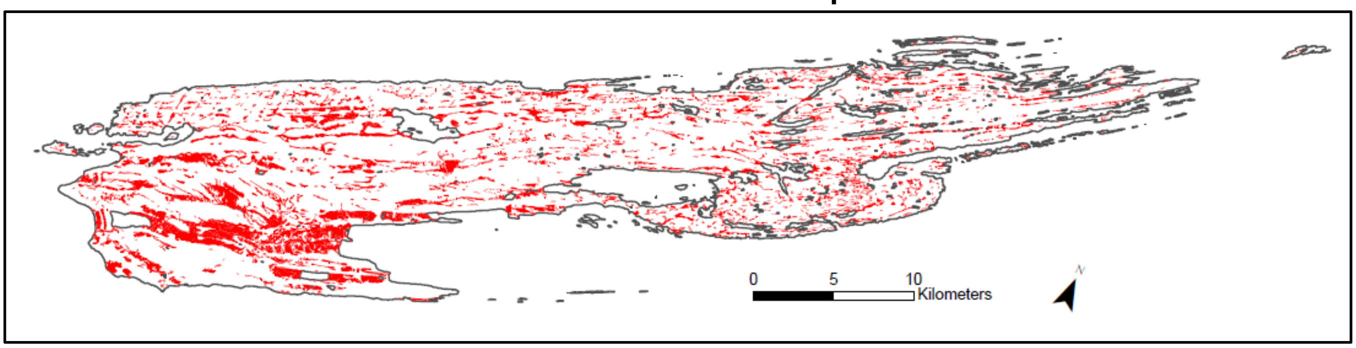


Figure 7. Distribution of the northern conifer and hardwood forest and shrub swamps ecological group on ISRO (15.54% of mapped vegetation cover). Map created using TNC GIS data (1999).

Northern conifer and hardwood forest and shrub swamps are a type of forested minerotrophic wetland, with common species that include northern white-cedar (*Thuja occidentalis*), black ash (*Fraxinus nigra*), and alder (*Alnus spp.*) (Kraft et al. 2010). Minerotrophic wetlands accumulate nutrients through surface- or groundwater-flows over rocks and minerals (Aber 2012). These ecosystems are often associated with headwater streams and areas of groundwater discharge (e.g. kettle depressions, peatland margins) on level or gently sloping terrain (Kraft et al. 2010; Kost et al. 2007).

Species composition is influenced by the groundwater seepage, which is rich in calcium and magnesium carbonates. Vegetation of these swamps is dominated by lowland hardwoods and conifers; the most common forest type in this group on Isle Royale is the cedar-speckled alder swamp, with scattered *Picea mariana* or *Abies balsamea*. Although the Tamarack–alder community type is rare, it does occur on the southwest part of island. Soils of these ecosystems are composed of a thin layer of organic soil (highly decomposed muck/peat) covering a poorly drained mineral substrate (Kraft et al. 2012; Kost et al. 2007), and species in these habitats typically germinate and establish on moss-covered hummocks or decaying logs (Kraft et al. 2012; Kost et al. 2007).

As previously stated, indirect effects under climate change include an increased risk of abiotic disturbances due to climate volatility (EFI 2007); patterns of abiotic disturbance largely shape Northern Conifer and Hardwood Forest and Shrub Swamps ecosystems. For example, windthrow creates canopy gaps and influences microtopography to facilitate a diversity of herbaceous flora. Additionally, wetland vegetation has adapted to specific water regimes (e.g. groundwater seepage, water table fluctuation, seasonal inundation and flooding), and disrupting hydrological patterns could lead to mismatches in phenology and moisture requirements (Short et al. 2016). Furthermore, plant community composition can change drastically as a result of wildfire that destroys the upper peat layers in soil (Rowe et al. 2017). Some species, such as black spruce, may benefit from low severity fire, as the combustion of peat facilitates seedling establishment (Rowe et al. 2017).

Great Lakes Rocky Shores

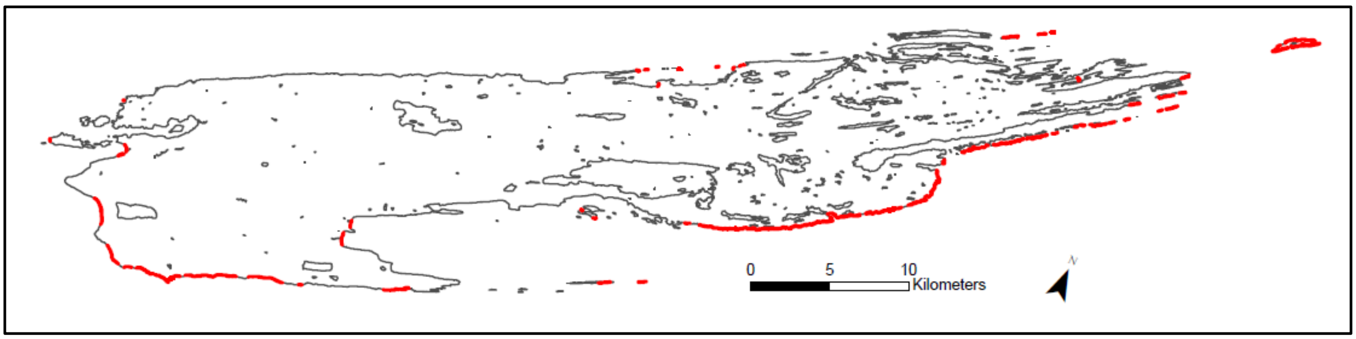


Figure 8. Distribution of the Great Lakes rocky shores ecological group on ISRO (0.26% of mapped vegetation cover). Extent has been exaggerated for improved visibility. Map created using TNC GIS data (1999).

The Great Lakes rocky shores ecosystem is characterized by areas where exposed bedrock dominates the shoreline. This bedrock can consist of metamorphic, sedimentary, or alkaline igneous rock (Albert 2003), and may be entirely solid rock or fragmented. The vegetation in these ecosystems can vary from non-vascular vegetation to shrubby and open-treed communities (Albert 2003). The vegetation type that is most common in this ecosystem is the Great Lakes alvars (Albert 2003), which are typically dominated by grasses and sedges, such as tufted hairgrass (*Deschampsia caespitosa*) and little green sedge (*Carex viridula*), as well as flowering forbs (e.g. Ontario lobelia (*Lobelia kalmii*)), and may also include sparse shrubs or trees (e.g. Northern white-cedar (*Thuja occidentalis*)).

The vegetation in this community is sparse, and those species that are able to grow are adapted to the harsh conditions and low nutrient levels. These species tend to be resilient to the changes that climate change can impose. The rocky shores ecology group may receive a slight benefit from climate change due to the increased volatility and incidents of disturbances that may maintain the open nature of these environments. However, many of these species tend to be at the southern extent of their distribution and may not cope with the warmer temperatures (WICCI 2017).

Regarding vulnerability to climate change, erosion is the primary concern facing the rocky shores ecosystem. Shorter winters may increase the frequency of freeze/thaw cycles, and thus result in increased bedrock fracturing (WICCI 2017). However, shorter winters may also reduce the ice scour of the shoreline (WICCI 2017). This reduction in ice scour may be offset by higher erosion rates due to increased severity of storms and waves throughout the year (WICCI 2017). A warming climate can increase evapotranspiration, which is predicted to decrease the water levels in Lake Superior by up to 0.2 meters by 2030 (Sea Grant 2005). Erosion rates may decrease due to shorter offshore wave height (Dickson 2007). Although there is a high uncertainty regarding the future water levels of Lake Superior, the rocky shores are particularly susceptible to increases in water level (WICCI 2017).

Rock Barrens

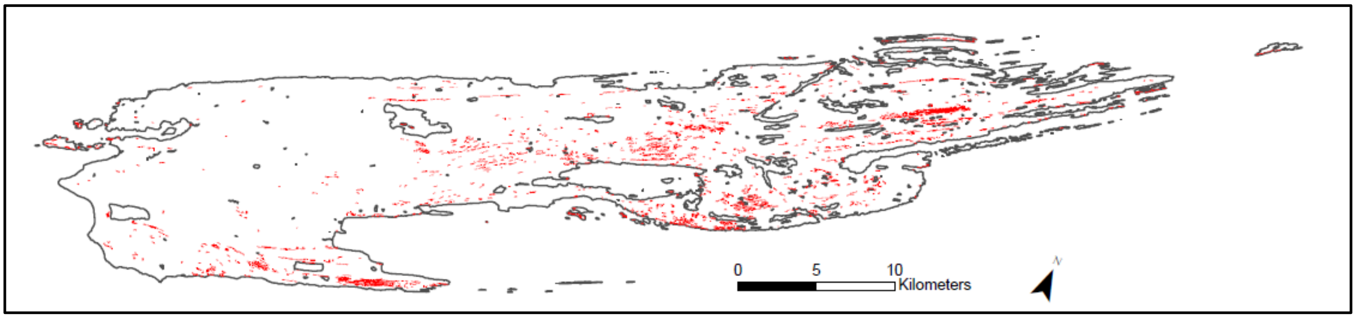


Figure 9. Distribution of the rock barrens ecological group on ISRO (3.34% of mapped vegetation cover). Map created using TNC GIS data (1999).

Rock Barrens are composed of basalt and gneissic or granitic barrens (Kraft et al. 2010). The basalt bedrock systems--found on Isle Royale--are characterized by exposed areas of bedrock and thin soils (Kraft et al. 2010). The plant communities are comprised of lichens, mosses, graminoids, shrub thickets, and scattered trees (Kraft et al. 2010). Common woody species include white spruce (*Picea glauca*), balsam fir (*Abies balsamea*), paper birch (*Betula papyrifera*), beaked hazelnut (*Corylus cornuta*), black hawthorn (*Crataegus douglasii*), and serviceberry (*Amelanchier* spp.). In ISRO, the rock barrens include one forested community, namely spruce-fir basalt bedrock glade, which is particularly susceptible to climate change. However, the rock barrens ecosystem as a whole has been very stable for the last 50 years (Kraft et al. 2010).

The primary vulnerability to this ecosystem is shoreline development and erosion (Kraft et al. 2010). As previously outlined, there is uncertainty regarding the future water levels of Lake Superior. Some studies indicate that the water levels may decline due to climate change, which would benefit shoreline development (Sea Grant 2005), while others suggest a water level increase, which would hinder shoreline development. As with other shoreline ecosystems, the shorter winters may increase the frequency of freeze/thaw cycles, which could increase bedrock fracturing (WICCI 2017). The shorter winters may also reduce the ice scour of the shoreline (WICCI 2017). This reduction in ice scour may be offset by an increased severity of storms and waves throughout the year that may increase erosion (WICCI 2017). The predicted volatility in precipitation regime and increased severity in precipitation events may cause further erosion (EFI 2007).

Cliffs and Talus

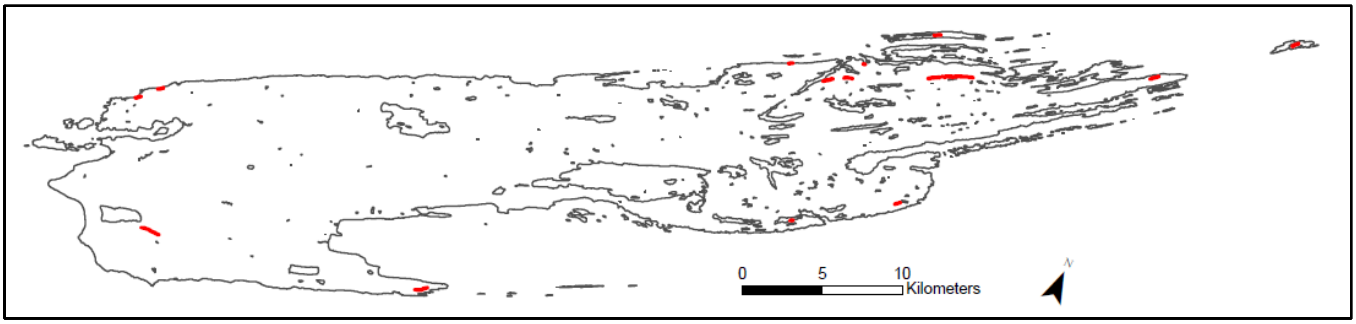


Figure 10. Distribution of the cliffs and talus ecological group on ISRO (0.05% of mapped vegetation cover). Extent has been exaggerated for improved visibility. Map created using TNC GIS data (1999).

Cliff and talus systems are typically harsh growing environments that require species with a wider geographic range. Vegetation able to grow in this environment includes ferns (e.g. smooth cliff brake (*Pellaea glabella*)) and herbaceous species (e.g. wild columbine (*Aquilegia canadensis*)) that are usually well adapted and resilient to changes climate change would impose. Cliffs and talus environments may even receive a slight benefit from climate change, because the increased volatility of the climate may maintain the open nature of these environments; increased occurrences of drought, flooding, windthrow, and ice damage can limit the spread of shade-producing trees or non-native invasives into these environments (WICCI 2017).

Similar to the rock barrens and Great Lakes rocky shores, the primary climate change risk associated with the cliff and talus ecological group is erosion. Extreme rain events may wash away soil and uproot shallow-rooted plants (WICCI 2017), which further destabilizes the soil. Shorter winters can increase the freeze/thaw cycles, which may increase bedrock fracturing (WICCI 2017). Furthermore, increased evapotranspiration from a warming climate is predicted to decrease the water levels in Lake Superior by up to 0.2 meters by 2030 (Sea Grant 2005). Erosion rates may be lowered as a result, as the offshore wave height is decreased (Dickson 2007).

Northern Dry Conifer (Hardwood) Forests and Woodlands

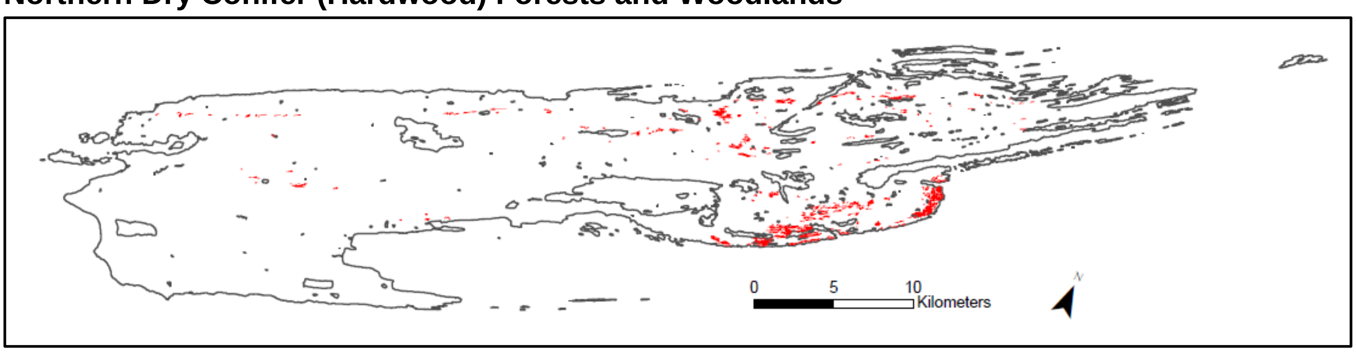


Figure 11. Distribution of the northern dry conifer (hardwood) forests and woodlands ecological group on ISRO (1.43% of mapped vegetation cover). Map created using TNC GIS data (1999).

Ecosystems falling within the northern dry conifer (hardwood) forests and woodlands category tend to be dominated by jack pine (*Pinus banksiana*), white pine (*Pinus strobus*), white spruce (*Picea glauca*), and black spruce (*Picea mariana*) in the canopy (TNC 1999). The understory species composition includes thimbleberry (*Rubus parviflorus*), blueberry (*Vaccinium angustifolium*), and common juniper (*Juniperus communis*), among other shrubs and herbaceous species. On Isle Royale, these systems can be found near the eastern coastlines of the island, as well as some interior ridges (Kraft et al. 2010).

As boreal species at the southernmost reaches of their limits, white and black spruce will likely become less competitive as average annual temperatures increase (Reich et al. 2015; Thompson et al. 2015). These species are likely to migrate northward due to climatic pressures such as heat and drought and eventually exit Isle Royale (Sanders & Grochowski 2013). *Pinus strobus*, as a temperate species currently growing at its northernmost limits, will likely experience a positive effect from warming temperatures.

Dry conifer forests, particularly boreal pine rocky woodlands, are also susceptible to fire, although TNC notes that fire may not be strictly necessary for jack pine to reproduce in these areas (TNC 1999). Projected increases in drought events, coupled with the ecosystem's history as a fire-established system, indicate an increased risk of higher-intensity crown fires, altering successional trajectories (TNC 1999). These species may be also at risk of increases in pest outbreaks, such as with the spruce budworm (Dale et al. 2001; Dukes et al. 2009). Increased spruce mortality during outbreaks may increase the amount of tinder available to feed ground fires, further increasing the potential for high-intensity fires in these systems. Increased frequency of fire could cause this system to transition towards a pine-dominated composition, as has been observed in a similarly rocky terrain in Quebec (Larocque et al. 2003).

Northern Mesic Conifer (Hardwood) Forests

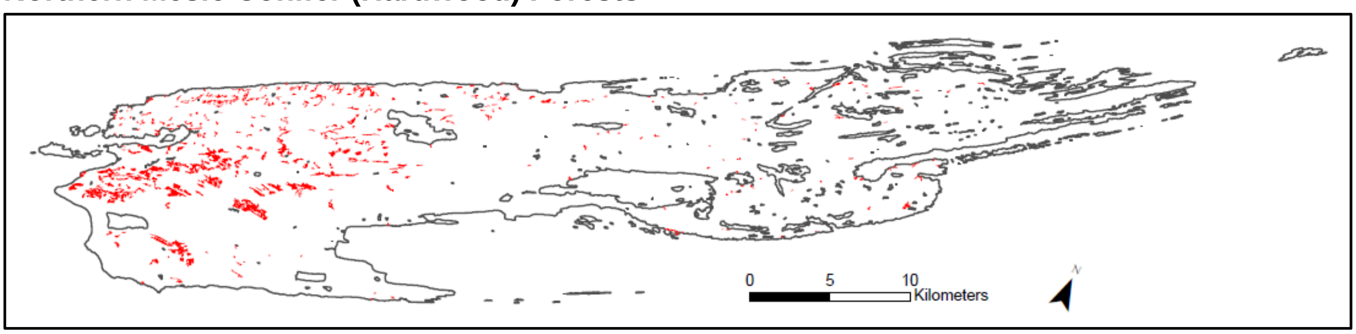


Figure 12. Distribution of the northern mesic conifer (hardwood) forests ecological group on ISRO (3.04% of mapped vegetation cover). Map created using TNC GIS data (1999).

On Isle Royale, northern mesic conifer (hardwood) forests are dominated by Northern white-cedar (*Thuja occidentalis*) in the canopy, often accompanied by yellow birch (*Betula alleghaniensis*), paper birch (*Betula papyrifera*), balsam fir (*Abies balsamea*), and/or white spruce (*Picea glauca*). As indicated in Figure 12 above, these ecosystems occur primarily on the southwestern end of the island (TNC 1999).

Species typically found in mesic environments, particularly white-cedar and yellow birch, may be susceptible to increased pressure from drought events and increases in summer temperatures (Thompson et al. 2015). While white-cedar in particular is not quite at the southernmost extremes of its distribution range on Isle Royale, its growth may still be subject to pressure from increased variation in seasonal patterns despite a projected longer growing season (Thompson et al. 2015). Even at the northernmost extent of its range an increase in temperature has led to an observed negative effect on growth, indicating that changes in precipitation patterns play a key role in the long-term competitiveness of the species (Housset et al. 2015). According to some models, white-cedar populations in the U.S., ISRO included, could decrease by at least 90% (Iverson & Prasad 2002). As the current dominant species in these ecosystems, a decrease in white-cedar's competitiveness would create opportunity for better-adapted species to outcompete and occupy a larger proportion of the canopy.

Northern Spruce-fir (Hardwood) Forests

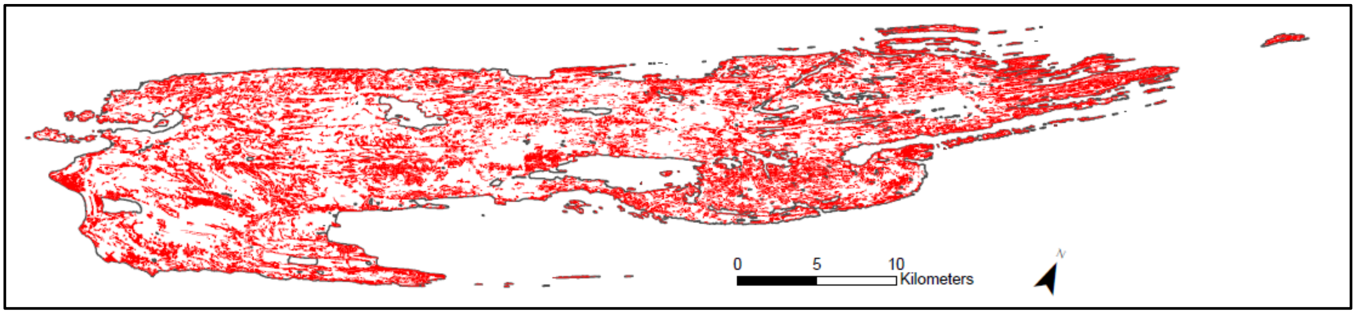


Figure 13. Distribution of the northern spruce-fir (hardwood) forests ecological group on ISRO (37.43% of mapped vegetation cover). Map created using TNC GIS data (1999).

Spruce-fir ecosystems are considered to be extremely vulnerable to climate change. Spruce-fir species are cold adapted boreal species that will become less competitive in warmer climates. The most dense spruce stands are found in cooler areas of Isle Royale, such as along the shoreline of Lake Superior or in cold pockets and depressions inland, although this community type is one of the most widespread in the park. The southern distribution of spruce-fir is limited by summer heat and drought (Iverson et al. 2001). Spruce growth rates declined when temperatures exceeded the optimum levels for photosynthesis (Janowiak et al. 2018).

Increased period between rain events, expected under climate change, adds to moisture stress and drought; these factors increase the risk of forest fires occurring on Isle Royale. Increased intensity of precipitation events in winter lead to heavier snow storms. Heavy loads of snow can damage trees and limit their growth. Extreme rain events in winter can waterlog the soil, reducing the anchorage for trees therefore increasing their susceptibility to windthrow damage (EFI 2007).

Biotic disturbances from pest outbreaks are also expected to increase with climate change. Outbreaks of pests such as the eastern spruce budworm can be precipitated by multiple years of favorable climate conditions for larval growth (Williams et al. 2000). The eastern spruce budworm larvae require warm, sunny, and dry conditions during spring for optimal growth and survival. These required conditions will become more common and lead to a northward range migration for this pest species (Williams et al. 2000).

Boreal Hardwood Forests and Woodlands

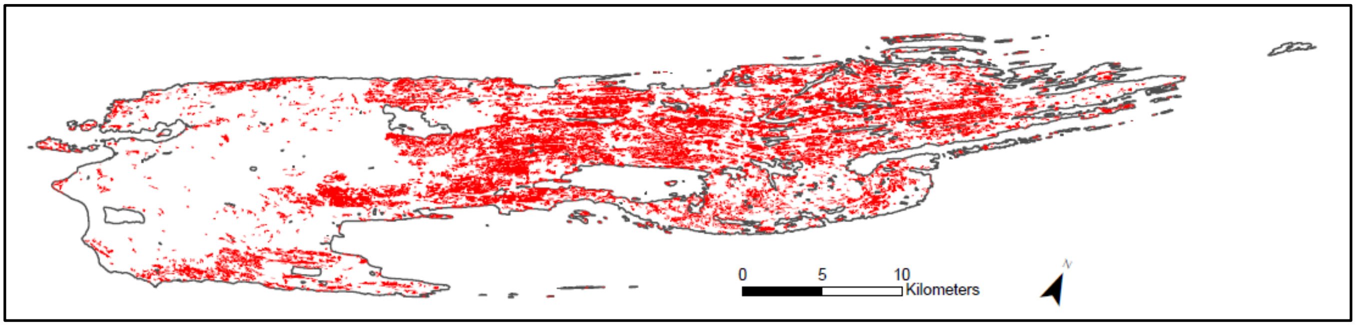


Figure 14. Distribution of the boreal hardwood forests and woodlands ecological group on ISRO (25.17% of mapped vegetation cover). Map created using TNC GIS data (1999).

Boreal hardwood species include those spruce and fir species typically found in spruce-fir communities, in addition to stands of trembling aspen (*Populus tremuloides*) and paper birch *(Betula papyrifera*). While an increase in temperature is expected to broadly increase the productivity of boreal hardwood species, growth will be limited by extremely high temperatures and lower nutrient availability (EFI 2007).

Higher temperatures without an increase in precipitation rates can lead to more frequent and prolonged droughts (EFI 2007). While the overall precipitation rates are predicted to increase, the number of rainy days is predicted to decrease. The number of heavy rain events is predicted to increase, while the intermediary conditions are drier and receive less precipitation (EFI 2007). Drier soil negatively impacts soil nutrient availability due to accelerated rates of nitrification (EFI 2007). Stress caused by lower nutrient and water availability leads to intense competition in the seedling layer. Boreal species in the short term are predicted to benefit from a warming climate through increased growth rates. However, boreal species on the southern edge of their distribution become less competitive compared to temperate species as alterations to temperature, moisture regime, CO_2, and radiation occur (Pedlar et al. 2017). Over the long-term, Boreal species distribution is expected to migrate northward.

Warmer winters may cause a decrease in soil freezing, which can reduce the soil anchorage for trees. This increases the tree susceptibility to snow damage and particularly windthrow damage (EFI 2007). The lack of anchorage is compounded with severe precipitation events that waterlog the soil and further weaken the soil anchorage for trees, increasing windthrow damage (EFI 2007). Boreal species on Isle Royale are distributed along the cooler coasts where the wind rate is higher. This makes Boreal species disproportionately more susceptible to windthrow damage than inland species.

Northern Hardwood Forests and Woodlands

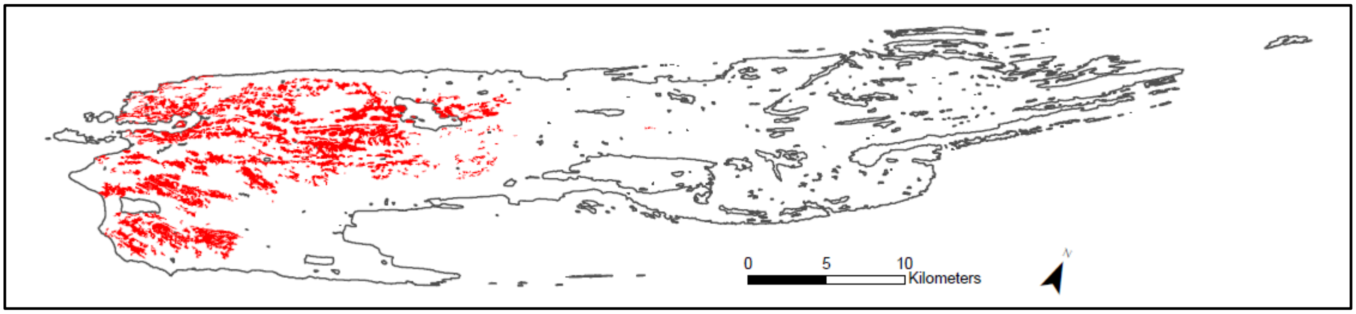


Figure 15. Distribution of the northern hardwood forests and woodlands ecological group on ISRO (9.71% of mapped vegetation cover). Map created using TNC GIS data (1999).

Northern hardwood forests in ISRO are dominated by both sugar maple (*Acer saccharum*) and yellow birch (*Betula alleghaniensis*), and are found primarily on the western third of the main island (TNC 1999). These particular forest types are very common on the nearby mainland in Minnesota, Michigan, and Wisconsin. Northern hardwood forest communities are generally found on mesic, well-drained soils. This is especially true on ISRO, as this forest type is only found in areas with older, more developed alluvial soils (Janke et al. 1978). The understory of northern hardwoods in ISRO is more species rich than the understories of other forests on the main island (Hansen et al. 1973). This understory, and constitutive soils, is less acidic and more nutrient rich than boreal forest understories because of higher quality leaf litter input (Pastor et al. 1993). These understories are comprised of species of ferns, small herbaceous plants including many spring ephemerals, and is represented heavily by woody recruits such as sugar maple seedlings (Hansen et al. 1973).

Northern hardwood forests on ISRO face both unique opportunities for expansion and potential novel stressors under climate change conditions. Many of the dominant tree species that compose these forests are near their northern latitudinal range edge on ISRO. Therefore, climatic distribution models that calculate future range distributions of these species predict a northward expansion of the community as temperatures rise in the near future (Fischelli et al 2014). This comes at the expense of receding boreal species failing to recruit in their historic ranges. It is important to note that while this pattern has been predicted by evaluating climatic variables, northern expansion of northern hardwood species, like sugar maple, may be limited by herbivory, lack of soil mutualists, seed predation, and lack of dispersers (Brown and Vellend 2014).

Most importantly, an increase in disturbance frequency (windthrow, fire, or pest outbreaks) may create canopy gaps in these forests. Given that many northern hardwood species are shade-tolerant, numerous forest gaps may allow for the colonization of fast growing, shade-intolerant species like some boreal hardwoods (trembling aspen and paper birch), as well as any potential invasive species in the area (Brown and Vellend 2014). Useful future research in this system should focus on the role that tree diversity and interactions with exotic earthworms will play with dominant species abundance (Handler et al. 2014).

Northern Shrublands

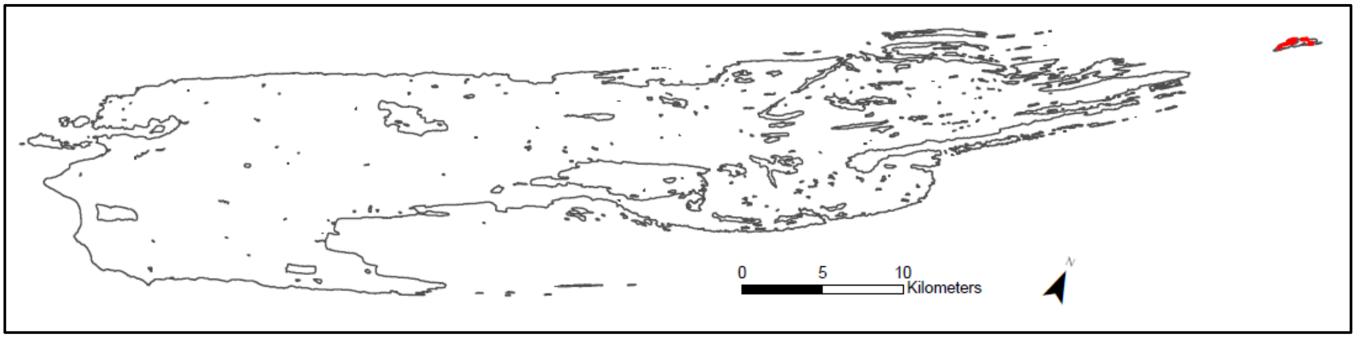


Figure 16. Distribution of the northern shrublands ecological group on ISRO (0.01% of mapped vegetation cover). Extent has been exaggerated for improved visibility. Map created using TNC GIS data (1999).

Northern shrubland ecosystems are open communities that usually emerge on bedrock outcrops near Lake Superior (Minnesota DNR 2018). The communities are usually inland from the wave-washed and ice-scoured zones (Minnesota DNR 2018). The vegetation is sparse and is composed of shrubs, herbaceous plants, graminoids, lichens, mosses, and occasionally scattered small trees (Minnesota DNR 2018). The trees that are able to grow in this environment usually are boreal species that have stunted growth. The species that grow in this community are generally exposed to greater environmental extremes than inland species. These extremes include rapid fluctuations of substrate temperature, higher rates of desiccation, limited nutrient availability, windthrow, and ice storms (Minnesota DNR 2018). The combination of higher winds and ice storms affect the shrubs and trees in this community, frequently leading to a stunting and "krummholz" growth forms ("Northern Bedrock Shrubland," 2018). Northern shrublands are a long-lived successional community that relies on forest fires to eliminate trees and consume soil organic deposits ("Northern Bedrock Shrubland," 2018).

Northern shrublands community species are drought tolerant and require fire disturbance to slow the eventual return of this ecosystem into a woodland ("Northern Bedrock Shrubland," 2018). Climate change is predicted to lead to warmer climates and a more volatile hydrological cycle. This will increase incidents of drought, which can increase the frequency of forest fires (EFI 2007). This would benefit the Northern shrubland species in slowing the transition into a woodland community.

Northern shrublands being open communities located near Lake Superior will be affected by increases in winter storms and windthrow damage ("Northern Bedrock Shrubland," 2018). Extreme precipitation events can saturate and waterlog the soils. Waterlogged soils can decrease the anchorage available to the shrubland species, causing increased windthrow damage (EFI 2007). Warmer and shorter winters can increase the freeze/thaw cycles, which may increase bedrock fracturing (WICCI 2017). Warmer winters may decrease the soil freezing, thereby diminishing the soil anchorage available for the shrubland species and making them more susceptible to snow damage (EFI 2007). Boreal shrubland species will experience short term increases in productivity (EFI 2007). However, boreal species are on their southern end of their distribution and may become less competitive compared to temperate species (Pedlar et al. 2017).

Semi-natural Meadows

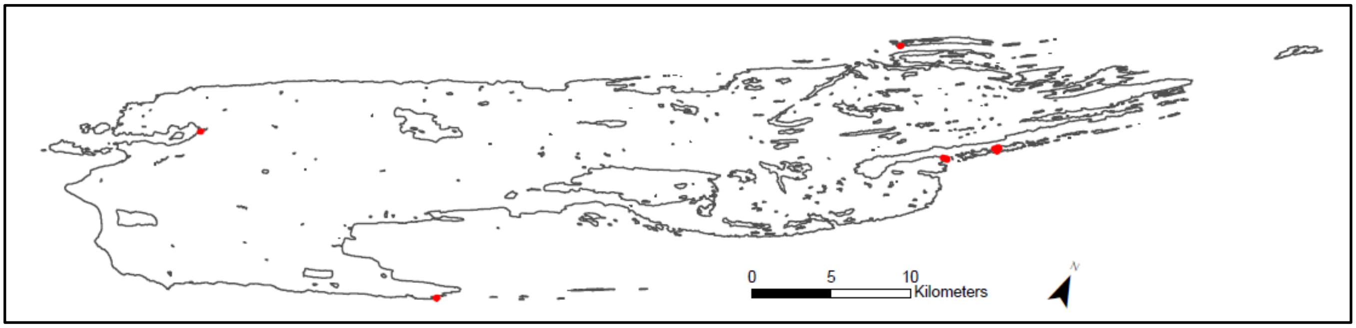


Figure 17. Distribution of the semi-natural meadows ecological group on ISRO (0.01% of mapped vegetation cover). Extent has been exaggerated for improved visibility. Map created using TNC GIS data (1999).

Semi-natural meadows are characterized as grasslands that are continuously managed. Semi-natural plant communities are defined as being composed of mostly indigenous species whose maintenance and development depend on direct human intervention (Linusson et al. 1998). These semi-natural pastures have exceptional biodiversity and act as a carbon sink. The soils are highly fertile organic soils, such as well decomposed peat (Kost et al. 2007). The vegetation types that dominate this ecosystem are grass (particularly Timothy grass (*Phleum pratense*) and bluejoint grass (*Calamagrostis canadensis*)) and sedge species, although woody plants do grow in this environment (Kost et al. 2007).

The primary vulnerability of this ecosystem to climate change is loss of biodiversity. A warming climate will disproportionately change the productivity of different species within this ecosystem. Increasing temperatures, moisture, CO_2, photoradiation, and increased nutrient levels can cause some species (especially some grasses) to become more competitive and displace the less competitive species (Pedlar et al. 2017). Long-term studies examining the loss of species diversity in semi-natural meadows have shown that the relative abundance of graminoids has been steadily increasing (Linusson et al. 1998). Loss of biodiversity and the emergence of dominant species, such as graminoid species, can potentially disrupt some ecological functions of semi-natural meadows, such as carbon sequestration. The loss of biodiversity may alter the productivity of this ecosystem.

Isle Royale Overall

The 14 vegetation communities of ISRO can be categorized into 4 distinct groups based upon similar responses to our examined climate change variables and scenarios (Figure 18 and 19). The wet group is composed of plant communities associated with some standing water, including: northern shrub/graminoid fens and bogs, rooted/floating aquatic marshes, wet meadows/marshes, and northern conifer and hardwood forest and shrub swamps. The rocky group is composed of plant communities found exclusively on bedrock or disturbed substrates and includes: Great Lakes rocky shores, rock barrens, cliffs and talus, semi-natural meadows, and northern shrublands. The boreal group is composed of communities dominated by boreal conifer and hardwood species and includes: northern dry conifer (hardwood) forests and woodlands, northern mesic conifer (hardwood) forests, northern spruce-fir (hardwood) forests, and boreal hardwood forests and woodlands. The last group is the temperate group, which is composed of the one community that supports temperate, deciduous species: northern hardwood forests and woodlands.

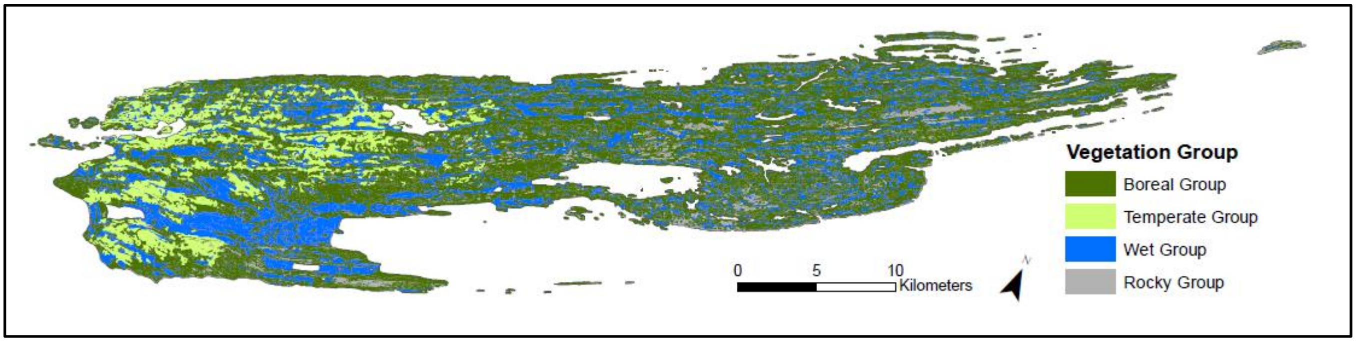


Figure 18. Distributions of four grouped communities (boreal, temperate, wet, and rocky) on ISRO. Map created using TNC GIS data (1999).

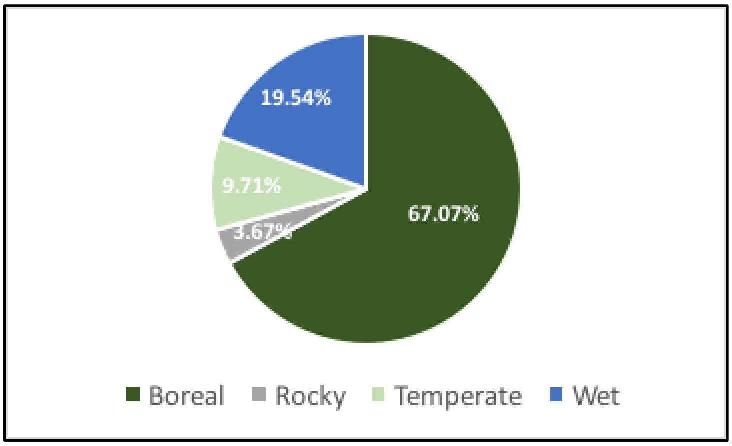


Figure 19. Percent of mapped vegetation cover occupied by each grouped community (boreal, temperate, wet, and rocky). Boreal communities occupy the greatest area of ISRO (67.07%), followed by wet (19.54%), temperate (9.71%), and rocky (3.67%) communities. Created using TNC data (1999).

Table 1 displays the response of each plant community to predicted climate change effects and stressors in the coming century, and Table 2 displays the community response within the context of each proposed climate change scenario from Fisichelli et al. (2013). Table 3 shows how the dominant species or plant groups within each community will respond to the four climate change scenarios. In general, the wet group, dominated by aquatic macrophytes, will be negatively affected by most climate change variables, especially increasing temperatures. Increased temperatures will increase the frequency and severity of droughts, and lower the water table within these communities. Since wet adapted plants, especially aquatic macrophytes, depend on standing water for growth, increased temperatures and evaporation may be extremely detrimental to wet community persistence. Since increasing temperatures is a prominent feature of each proposed climate change scenario, aquatic communities fare poorly in each case.

The rocky plant community group will have mixed responses to climate change. While the lack of winter ice and snow cover, as well as increased temperatures, are predicted to increase the growth and reduce the physical damage of the plants associated with bedrock, shores, or cliffs, the increase in wind and extreme events may increase plant desiccation. Rocky communities should be successful in scenarios where these disturbances are infrequent, but are likely to suffer in scenarios where they are common (i.e. the summer drought, wind, and fire scenario). This is also true of beaked hazelnut (*Corylus cornuta*), the dominant shrub in this group, and of the lichens and mosses that successfully colonize bare rock. Graminoids represent an alternate response to climate change stressors, and they are forecasted to increase in extent for all climate scenarios.

The species of the boreal community group (spruces, balsam fir, pines, trembling aspen, and paper birch) are predicted to benefit only from increased precipitation. However, the species within these groups are sensitive to thermal stress from increased temperatures, as well as frequent disturbances such as fire and insect outbreaks. Not surprisingly, the boreal group is expected to perform poorly in each proposed climate change scenario. Because Isle Royale is located at the southern extent of the boreal biome, these species are required to seek out specific microclimates for refuge, such as along the cooler, lower nutrient shorelines and in inland cold pockets and depressions. The warming climate will exert added pressure and reduce the distribution of these cold refuges. Additionally, the frequency of disturbance along the shoreline is expected to increase. The incidents of storms and extreme weather events will affect shore ecosystems more than inland ecosystems, and are thus predicted to decrease the diversity and richness of boreal species, such as balsam fir, trembling aspen, and black spruce (Table 3). This is also supported by other assessments focusing on the upper Midwest (Handler et al. 2014).

Lastly, the temperate plant community group, represented by northern hardwoods, is predicted to increase in extent over the next century. This is primarily due to the capacity of northern hardwood species (sugar maple and yellow birch) to capitalize on increased growth via higher temperatures, higher precipitation, and a longer growing season, which makes them more competitive than their boreal species counterparts. Overall, we predict a net increase of temperate forest cover on ISRO, with a corresponding decrease in boreal forest cover over the next century when considering climate change interactions. This may be limited by biotic drivers such as herbivory, but these interactions are less understood. Additionally, while the community as a whole may benefit or suffer from climate change, the composition of these forests may dramatically change via differential success of component species within these communities (Figure 20).

Table 1. Change in extent of the vegetation communities on ISRO given the most common predicted climate change variables. + indicates an increase in cover, and a – indicates a decrease in predicted cover over the next century. A +/- indicates that the community will have a mixed response.

	Projected Changes in Climate or Physical Variables as a Result of Climate Change by 2100					
Vegetation Community	Increasing Temperatures	Increasing Precipitation	Decreasing Snowfall	Longer Growing Season	Increasing Wind	Increasing Extreme Weather Events (Increased temperature and precipitation)
Northern shrub/graminoid fens and bogs	-	+	-	-	-	+/-
Rooted/floating aquatic marshes	-	+	-	-	-	+/-
Wet meadows/marshes	-	+	-	-	-	+/-
Northern conifer and hardwood forest and shrub swamps	-	+	-	+	-	+/-
Great Lakes rocky shores	+	-	+	-	-	-
Rock barrens	+	-	+	-	-	-
Cliffs and talus	+	-	+	-	-	-
Northern dry conifer (hardwood) forests and woodlands	+	+	-	+	-	-
Northern mesic conifer (hardwood) forests	+	+	-	+	-	-
Northern spruce-fir (hardwood) forests	-	+	-	+	-	-
Boreal hardwood forests and woodlands	-	+	-	+	-	-
Northern hardwood forests and woodlands	+	+	-	+	-	-
Northern shrublands	+	+	+	+	-	-
Semi-natural meadows	-	-	-	-	+/-	-

Table 2. Change in extent of the vegetation communities on ISRO given the four climate change scenarios discussed by Fisichelli et al. (2013). + indicates an increase in cover, and a – indicates a decrease in predicted cover over the next century. A +/- indicates that the community will have a mixed response.

	Climate Change Scenarios			
Vegetation Community	Least Change	Summer Drought, Wind, and Fire	Warmer than Duluth	Isle Savanna
Northern shrub/graminoid fens and bogs	-	-	-	-
Rooted/floating aquatic marshes	+/-	-	-	-
Wet meadows/marshes	+/-	-	-	-
Northern conifer and hardwood forest and shrub swamps	-	-	-	-
Great Lakes rocky shores	+	-	+	+
Rock barrens	+	-	+	+
Cliffs and talus	+	-	+	+
Northern dry conifer (hardwood) forests and woodlands	+	+	+	-
Northern mesic conifer (hardwood) forests	+/-	-	-	-
Northern spruce-fir (hardwood) forests	-	-	-	-
Boreal hardwood forests and woodlands	-	-	-	-
Northern hardwood forests and woodlands	+	+	+	-
Northern shrublands	+	+	+	+
Semi-natural meadows	+	+	+	+

Table 3. Change in extent of the dominant plant species and groups within each vegetation community on ISRO given the four climate change scenarios discussed by Fisichelli et al. (2013). + indicates an increase in cover, and a – indicates a decrease in predicted cover over the next century. A +/- indicates that the community will have a mixed response.

Dominant Plants	Vegetation Community	Major Stressors	Least Change	Summer Drought, Wind, and Fire	Warmer than Duluth	Isle Savanna
Sugar maple	Northern hardwood forests and woodlands	Increase in extreme events	+	+	+	-
Yellow birch	Northern hardwood forests and woodlands	Increase in extreme events	+	+	+	-
Trembling aspen	Boreal hardwood forests and woodlands	Increase in temperature; heavy browse	-	-	-	-
Paper birch	Boreal hardwood forests and woodlands	Increase in temperature; heavy browse	-	-	-	-
White spruce	Northern spruce-fir (hardwood) forests	Increase in temperature	-	-	-	-
Black spruce	Northern spruce-fir (hardwood) forests	Increase in temperature	-	-	-	-
Balsam fir	Northern spruce-fir (hardwood) forests	Increase in temperature; heavy browse	-	-	-	-
Northern white-cedar	Northern conifer and hardwood forest and shrub swamps	Increase in temperature	-	-	-	-
Black ash	Northern conifer and hardwood forest and shrub swamps	Increase in temperature; disease	-	-	-	-
Beaked hazelnut	Northern shrublands	Increase in extreme events	+	+	+	+
White pine	Northern mesic conifer (hardwood) forests	Increase in extreme events; disease	+/-	-	-	-
Jack pine	Northern dry conifer (hardwood) forests and woodlands	Increase in extreme events	+	+	+	-
Aquatic macrophytes	Rooted/floating aquatic marshes	Increase in temperature	+/-	-	-	-
Graminoids	Wet meadows/marshes; Semi-natural meadows	Increase in extreme events	+	+	+	+
Sphagnum	Northern shrub/graminoid fens and bogs	Increase in temperature	-	-	-	-
Lichens and Mosses	Great Lakes Rocky Shores; Rock barrens; Cliffs and talus	Increase in extreme events and wind	+/-	+/-	-	-

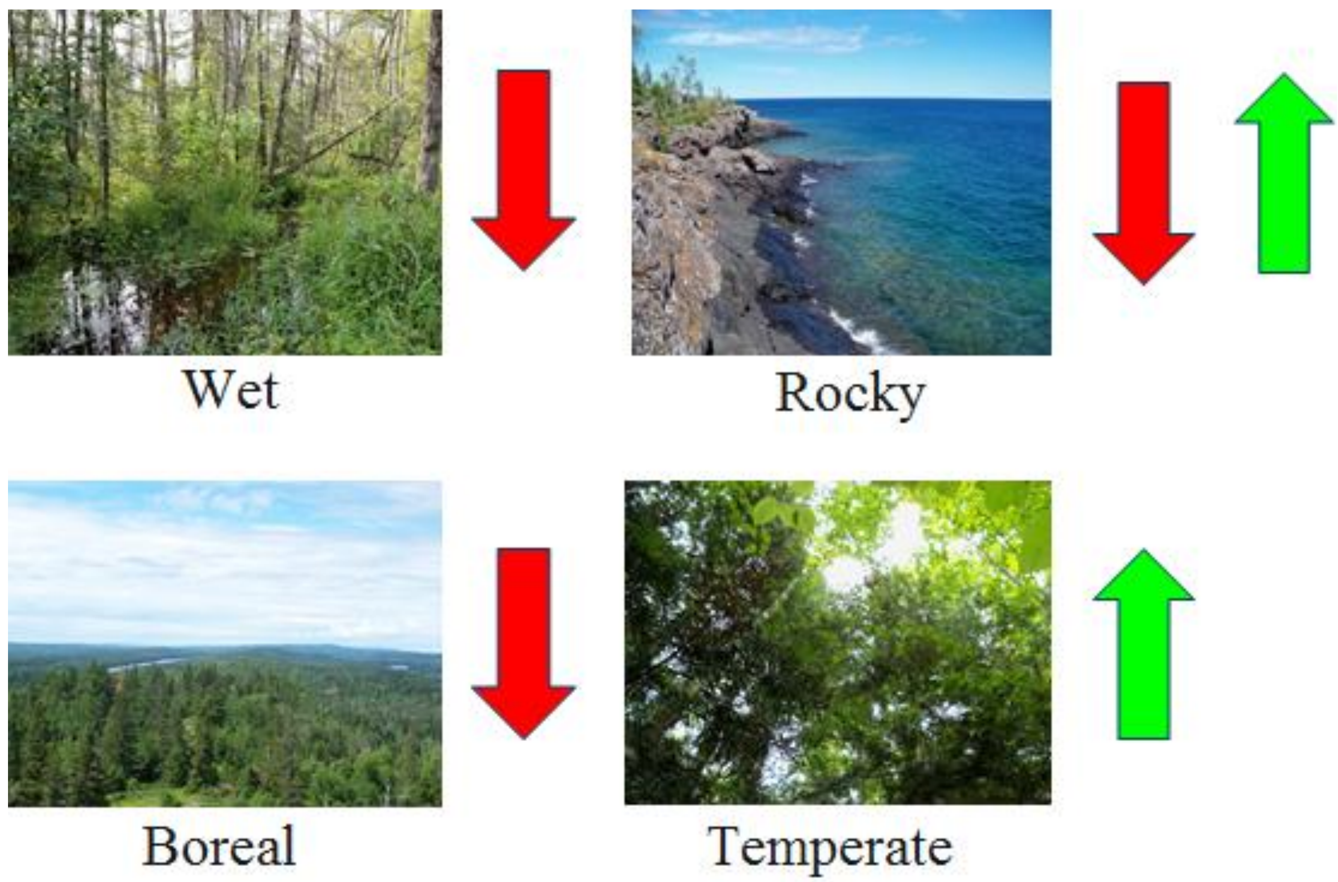


Figure 20. Illustration of general community level trends in response to climate change for the four highlighted community categories.

Conclusions

As demonstrated in the findings of this review, the ecosystems in ISRO have experienced and will continue to experience individualized benefits and adversities due to climate change. These impacts are reflected at multiple scales, as individual species (Table 3) and ecosystem groups (Table 1 and 2) will respond differently to altering conditions. Climate change in the upper Midwest is forecasted to increase temperatures, increase precipitation in the form of rainfall, decrease snowfall, increase the length of frost-free growing season, decrease the extent of ice cover in Lake Superior, and increase the frequency of extreme storm events, causing increased frequency and severity of disturbances like fire and windthrow. This may also increase the frequency of other types of disturbances such as insect outbreaks.

We isolated effects of these climate change stressors on each of the identified 14 plant communities. We then placed these communities into 4 broad categories: wet adapted communities, rocky talus or shoreline communities, boreal forests, and temperate forests. The literature suggests that wet communities will respond poorly to climate change, especially if increased temperatures greatly increases evapotranspiration. Dry rocky communities will have mixed responses: less ice cover will cause less damage to these plants; though an increase in disturbance from wind may also cause damage. Southern temperate forests will likely increase in extent with rising temperatures, while northern boreal forests will likely decrease in extent. However, we still need to understand how biotic drivers, like herbivory, will influence these dynamics. It is also important to note that individual species will have varying responses to climate change and that plant communities as a whole may not respond or migrate synchronously with their constitutive communities due to increased stress.

We present this assessment as a useful guide for ISRO natural resource managers; however we stress that these findings are derived from a qualitative review of the relevant literature, and should not replace further empirical study to fully evaluate plant community level responses to climate change. We hope that this document will help inform management decisions aimed at preserving community-level biotic function and conserving species within at-risk plant communities.

Literature Cited

Aber, J.S. (2012). Wetland definitions and classification. Emporia State University. http://academic.emporia.edu.

Albert, D. (2003). Ecological System Comprehensive Report. NatureServe Explorer. Retrieved from http://explorer.natureserve.org.

Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., & Gonzalez, P. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management, 259(4), 660-684.

Bonan, G. B., & Shugart, H. H. (1989). Environmental factors and ecological processes in boreal forests. Annual review of ecology and systematics, 20(1), 1-28.

Brandner, T. A., Peterson, R. O., & Risenhoover, K. L. (1990). Balsam fir on Isle Royale: effects of moose herbivory and population density. Ecology, 71(1), 155-164.

Brown, C. D., & Vellend, M. (2014). Non-climatic constraints on upper elevational plant range expansion under climate change. Proceedings of the Royal Society of London B: Biological Sciences, 281(1794), 20141779.

Carlisle, D. (2000). Summary of recent analyses of Isle Royale aquatic resources. Unpublished report to Isle Royale National Park, Houghton, Michigan.

Dickson, M. E., Walkden, M. J., & Hall, J. W. (2007). Systemic impacts of climate change on an eroding coastal region over the twenty-first century. Climatic Change, 84(2), 141-166.

Dukes, J. S., J. Pontius, D. A. Orwig, J. R. Garnas, V. L. Rodgers, N. J. Brazee, B. J. Cooke, K. A. Theoharides, E. E. Stange, R. A. Harrington, J. G. Ehrenfeld, J. Gurevitch, M. Lerdau, K. Stinson, R. Wick, and M. P. Ayres. (2009). Responses of insect pests, pathogens and invasive species to climate change in the forests of northeastern North America: What can we predict? Canadian Journal of Forest Research 39:231-248.

Evans, P., & Brown, C. D. (2017). The boreal–temperate forest ecotone response to climate change. Environmental Reviews, 25(4), 423-431.

European Forest Institute (EFI). (2007). Forest typology and expected climatic changes in different bioclimatic zones.

Fisichelli, N., Hoffman, C. H., Welling, L., Briley, L., & Rood, R. (2013). Using climate change scenarios to explore management at Isle Royale National Park: January 2013 workshop report.Natural Resource Report NPS/NRSS/CCRP/NRR—2013/714. National Park Service, Fort Collins, Colorado.

Fisichelli, N. A., Frelich, L. E., & Reich, P. B. (2014). Temperate tree expansion into adjacent boreal forest patches facilitated by warmer temperatures. Ecography, 37(2), 152-161.

Flakne, R. (2003). The Holocene vegetation history of Isle Royale National Park, Michigan, USA. Canadian Journal of Forest Research, 33(6), 1144-1166.

Goldblum, D., & Rigg, L. S. (2010). The deciduous forest–boreal forest ecotone. Geography Compass, 4(7), 701-717.

Handler, S., Duveneck, M. J., Iverson, L., Peters, E., Scheller, R. M., Wythers, K. R., ... & Swanston, C. (2014). Michigan forest ecosystem vulnerability assessment and synthesis: a report from the Northwoods Climate Change Response Framework project. *Gen. Tech. Rep. NRS-129. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 229 p.*, *129*, 1-229.

Hansen, H., L. Krefting, and V. Kurmis. (1973). The forest of Isle Royale in relation to fire history and wildlife. Technical Bulletin 294, University of Minnesota Agricultural Experiment Station, St. Paul, Minnesota.

HilleRisLambers, J., Harsch, M. A., Ettinger, A. K., Ford, K. R., & Theobald, E. J. (2013). How will biotic interactions influence climate change–induced range shifts?. Annals of the New York Academy of Sciences, 1297(1), 112-125.

Housset, J.M., Girardin, M.P., Baconnet, M., Carcaillet, C., & Bergeron, Y. (2015). Unexpected warming-induced growth decline in Thuja occidentalis at its northern limits in North America. Journal of Biogeography, 42(7), 1233-1245.

Iverson, L. R., & Prasad, A. M. (2001). Potential Changes in Tree Species Richness and Forest Community Types following Climate Change. Ecosystems, 4(3), 186-199.

Iverson, L. R., & Prasad, A. M. (2002). Potential redistribution of tree species habitat under five climate change scenarios in the eastern US. Forest Ecology and Management, 155(1-3), 205-222.

Janowiak, M. K., D'Amato, A. W., Swanston, C. W., Iverson, L., Thompson, F. R., Dijak, W. D., . . . Templer, P. (2018). New England and northern New York forest ecosystem vulnerability assessment and synthesis: a report from the New England Climate Change Response Framework project. United States Department of Agriculture.

Kool, A., & Heijmans, M. M. (2009). Dwarf shrubs are stronger competitors than graminoid species at high nutrient supply in peat bogs. Plant Ecology, 204(1), 125-134.

Kost, M. A., Albert, D. A., Cohen, J. G, Slaughter, B. S., Schillo, R. K., Weber, C. R., and Chapman., K. A. (2007). Natural Communities of Michigan: Classification and Description. Michigan Natural Features Inventory, Report No. 2007-21, Lansing, MI.

Kraft, G. J., Mechenich, D.J., Mechenich, C., Cook, J. E., & Seiler, S.M. (2010). Assessment of natural resource conditions: Isle Royale National Park. Natural Resource Report NPS/NRPC/WRD/NRR—2010/237. National Park Service, Fort Collins, Colorado.

Larocque, I., Bergeron, Y., Campbell, I.D., & Bradshaw, R. H. W. (2003). Fire-induced decrease in forest cover on a small rock outcrop in the Abitibi region of Quebec, Canada. Ecoscience, 10(4), 515-524.

Linusson, A. C., Berlin, G. A. I., & Olsson, E. G. A. (1998). Reduced community diversity in semi-natural meadows in southern Sweden, 1965–1990. Plant Ecology, 136(1), 77-94.

McInnes, P. F., Naiman, R. J., Pastor, J., & Cohen, Y. (1992). Effects of moose browsing on vegetation and litter of the boreal forest, Isle Royale, Michigan, USA. Ecology, 73(6), 2059-2075.

Meeker, J., Harris, A., Hofman, V., Judziewicz, E., & Marr., J. (2007). Assessment of wetland communities at Isle Royale National Park. National Park Service, Great Lakes Inventory and Monitoring Network Report GLKN/2007/04. NPS, Ashland, Wisconsin.

Minnesota DNR. (2018). ROn23 Northern Bedrock Shrubland factsheet. Retrieved from https://files.dnr.state.mn.us/natural_resources/npc/rock_outcrop/ron23.pdf.

Nelson, M. P., Peterson, R. O., & Vucetich, J. A. (2008). The Isle Royale wolf–moose project: Fifty years of challenge and insight. In The George Wright Forum, 5(2), 98-113.

Pastor, J., Naiman, R.J., & Dewey, B. (1987). A hypothesis of the effects of moose and beaver foraging on soil nitrogen and carbon dynamics, Isle Royale. Alces 23: 107-124.

Pastor, J., Dewey, B., Naiman, R. J., McInnes, P. F., & Cohen, Y. (1993). Moose browsing and soil fertility in the boreal forests of Isle Royale National Park. Ecology, 74(2), 467-480.

Pastor, J., Dewey, B., Moen, R., Mladenoff, D. J., White, M., & Cohen, Y. (1998). Spatial patterns in the moose–forest–soil ecosystem on Isle Royale, Michigan, USA. Ecological Applications, 8(2), 411-424.

Pedlar, J. H., & Mckenney, D. W. (2017). Assessing the anticipated growth response of northern conifer populations to a warming climate. Scientific Reports,7, 43881. doi:10.1038/srep43881

Post, E., & Pedersen, C. (2008). Opposing plant community responses to warming with and without herbivores. Proceedings of the National Academy of Sciences, 105(34), 12353-12358.

Rotter, M. C., & Rebertus, A. J. (2014). Plant community development of Isle Royale's moose-spruce savannas. Botany, 93(2), 75-90.

Rowe, E. R., D'Arnato, A. W., Palik, B. J., Almendinger, J. C. (2017). Early response of ground layer plant communities to wildfire and harvesting disturbance in forested peatland ecosystems in northern Minnesota, USA. Forest Ecology & Management, 398, 140-152.

Sanders, S., & Grochowski, J. (2013). The Forests of Isle Royale National Park: Can We Preserve This Pristine Wilderness in the Face of Climate Change? Natural Areas Journal, 33(1), 66-77.

Short, F. T., Kosten, S., Morgan, P. A., Malone, S., & Moore, G. E. (2016). Impacts of climate change on submerged and emergent wetland plants. Aquatic Botany, 135, 3-17.

Thompson, R. S., Anderson, K. H., Pelltier, R. T., Strickland, L. E., Shafer, S. L., Bartlein, P. J., and McFadden, A. K. (2015). Atlas of relations between climatic parameters and distributions of important trees and shrubs in North America—Revisions for all taxa from the United States and Canada and new taxa from the western United States: U.S. Geological Survey Professional Paper 1650–G, http://dx.doi.org/10.3133/pp1650G.

TNC (The Nature Conservancy). (1999). USGS-NPS Vegetation Mapping Program: classification of the vegetation of Isle Royale National Park. The Nature Conservancy. Minneapolis, Minnesota. Online. (http://biology.usgs.gov/npsveg/isro/).

USFS, Previant, W. J., Nagel, L. M., Pugh, S. A., & Woodall, C. W. (2012). Forest Resources of Isle Royale National Park 2010. Resour. Bull. NRS-73. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 24 p., 73, 1-24.

USGS (United States Geological Survey). (2000). Isle Royale National Park spatial vegetation data. USGS, Denver, Colorado. Online. (http://biology.usgs.gov/npsveg/isro/index.html#geospatial_veg_info).

Vucetich, J. A., Peterson, R. O., & Schaefer, C. L. (2002). The effect of prey and predator densities on wolf predation. Ecology, 83(11), 3003-3013.

Vucetich, J. A., & Peterson, R. O. (2004). The influence of top–down, bottom–up and abiotic factors on the moose (*Alces alces*) population of Isle Royale. Proceedings of the Royal Society of London B: Biological Sciences, 271(1535), 183-189.

Williams, D. W., Long, R. P., Wargo, P. M., & Liebhold, A. M. (2000). Effects of Climate Change on Forest Insect and Disease Outbreaks. Ecological Studies Responses of Northern U.S. Forests to Environmental Change,455-494. doi:10.1007/978-1-4612-1256-0_13

Wisconsin Department of Natural Resources. (2015). Wisconsin Wildlife Action Plan [WWAP]; Threats/Issues and Conservation Actions - Northern Forest Group. Wisconsin Department of Natural Resources, Madison, WI. Retrieved from https://dnr.wi.gov/topic/endangeredResources/documents/norForConservationActions.pdf

Wisconsin Initiative on Change Impacts [WICCI]. (2017). Climate Vulnerability Assessments for Plant Communities of Wisconsin. Wisconsin Initiative on Climate Change Impacts, Madison, WI. https://www.wicci.wisc.edu/resources/Dry_Cliff_CCVA.pdf

Wisconsin Sea Grant. Anticipated future lake levels on Superior, Michigan and Huron due to climate change. (2005). Retrieved from http://www.seagrant.wisc.edu.

Wu, J., & Roulet, N. T. (2014). Climate change reduces the capacity of northern peatlands to absorb the atmospheric carbon dioxide: The different responses of bogs and fens. Global Biogeochemical Cycles, 28(10), 1005-1024. doi:10.1002/2014gb004845.