Disturbances in North American boreal forest and tundra: impacts, interactions, and responses

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Abstract

Ecosystems in the North American Arctic-Boreal Zone (ABZ) experience a diverse set of disturbances associated with wildfire, permafrost dynamics, geomorphic processes, insect outbreaks and pathogens, extreme weather events, and human activity. Climate warming in the ABZ is occurring at over twice the rate of the global average, and as a result the extent, frequency, and severity of these disturbances are increasing rapidly. Disturbances in the ABZ span a wide gradient of spatiotemporal scales and have varying impacts on ecosystem properties and function. However, many ABZ disturbances are relatively understudied and have different sensitivities to climate and trajectories of recovery, resulting in considerable uncertainty in the impacts of climate warming and human land use on ABZ vegetation dynamics and in the interactions between disturbance types. Here we review the current knowledge of ABZ disturbances and their precursors, ecosystem impacts, temporal frequencies, spatial extents, and severity. We also summarize current knowledge of interactions and feedbacks among ABZ disturbances and characterize typical trajectories of vegetation loss and recovery in response to ecosystem disturbance using satellite time-series. We conclude with a summary of critical data and knowledge gaps and identify priorities for future study.

Keywords: high-latitude, vegetation, boreal forest, arctic tundra, climate change, disturbance, permafrost

1. Introduction

In the North American Arctic-Boreal Zone (ABZ), climate change and human activity are rapidly and extensively reshaping vegetation dynamics via a range of disturbance processes, resulting in considerable uncertainty in the fate of these ecosystems (Shaw *et al* 2021). Many disturbances (i.e., an event that alters ecosystem composition, structure, function, or the physical environment, Pickett and White 1985) trigger a transient reduction and gradual recovery of vegetation cover and ecosystem function (Liu *et al* 2011, Li *et al* 2021), although there is high variability in the nature and pace of these changes depending on the type and severity of disturbance (Jorgenson *et al* 2015, Gaglioti *et al* 2021) (Fig. 1). Climate warming is occurring in the ABZ at more than twice the global average rate (Price *et* *al* 2013, Smith *et al* 2019, Chylek *et al* 2022, Rantanen *et al* 2022), and many disturbance processes are highly sensitive to climate. Consequently, the impact of climate change via disturbance on ABZ vegetation dynamics is expected to increase over the next century (Price *et al* 2013, Gauthier *et al* 2015, Smith *et al* 2019, Bush and Lemmen 2019).

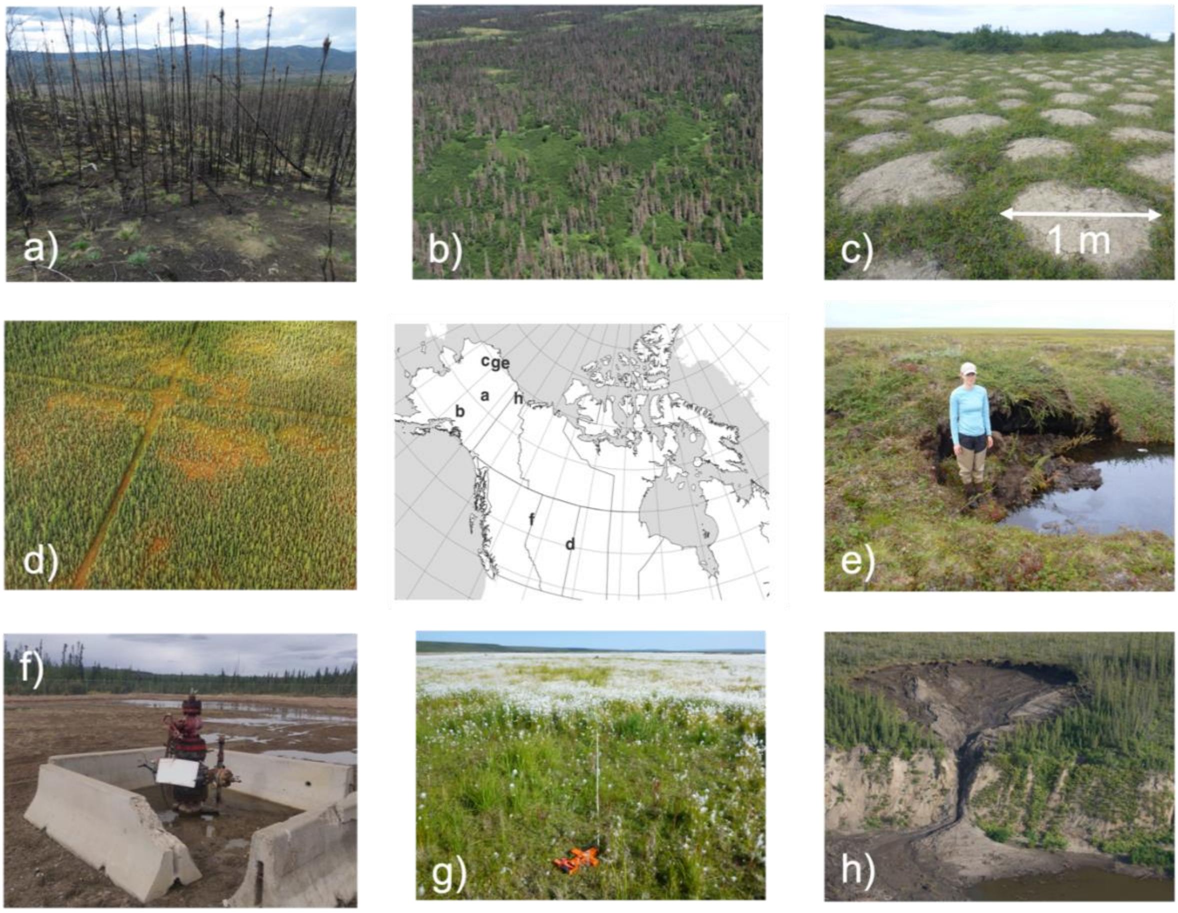

Figure 1. Examples of disturbances and successional responses in North American Arctic and boreal forest ecosystems. a) Burned (2020) upland black spruce forest in early succession, Interior Alaska; b) spruce beetle infestation in 2016, south-central Alaska, credit Bruce Cook; c) non-sorted circles arising from cryoturbation, Alaska North Slope; d) seismic line disturbance cutting across a treed peatland, northern Alberta, Canada; e) thermokarst after ice-wedge degradation, Alaska North Slope; f) suspended oil and gas well, drilled in 2006, north-eastern British Columbia, Canada; g) recently drained lake basin in early succession, Alaska North Slope; h) thaw slump, Old Crow Flats, Yukon, Canada.

Disturbance-driven loss and subsequent recovery of vegetation partly explain widespread trends in satellite-observed vegetation indices (i.e., “greening” and “browning”) within the North American ABZ **(**Wang and Friedl 2019, Sulla-Menashe *et al* 2018, Ju and

Masek 2016). Large-scale greening trends across the ABZ are complex (Myers-Smith *et al* 2020), but have generally been interpreted as an increase in ecosystem productivity driven by climatic warming and recovery from disturbance (Bhatt *et al* 2010, Berner *et al* 2020). Meanwhile, areas of browning are generally attributed to vegetation stress from disturbances such as fires, insect outbreaks, warming-induced drought, and increased surface water associated with permafrost degradation (Goetz *et al* 2005, Berner and Goetz 2022, Verbyla 2011, Shur and Jorgenson 2007). Many of these disturbances are increasing in their extent, frequency, and/or severity because of climatic changes and increasing anthropogenic pressures (Jorgenson *et al* 2006, Baltzer *et al* 2021). Understanding the net impact of climate change and its effects on different disturbance regimes is critical for forecasting future ABZ composition, dynamics, ecosystem services, and potential management responses.

As in many other ecosystems, fires have dramatic and extensive impacts on vegetation cover and carbon dynamics in the ABZ, and exceptional warming in this region is intensifying fire regimes (Soja *et al* 2007, Veraverbeke *et al* 2017, Kasischke *et al* 2010, Whitman *et al* 2022, McCarty *et al* 2021). However, the unique characteristics of ABZ ecosystems result in additional types of disturbances that lack analogs in tropical and temperate ecosystems. The wide extent of permafrost (i.e., perennially frozen ground; Gruber 2012) that underlies large parts of the northern high-latitudes makes these ecosystems vulnerable to a unique set of other disturbances (Shur and Jorgenson 2007). For example, thawing permafrost causes ground surface subsidence that can induce persistent changes in hydrology, vegetation, and microtopography in ABZ landscapes with high ground-ice contents (Jones *et al* 2015, Farquharson *et al* 2019, Swanson 2021, Carpino *et al* 2018, Grosse *et al* 2011). Exceptional warming in the ABZ also makes high-latitude forests vulnerable to increasing incidences of drought and insect outbreaks (Volney and Fleming 2000, Hogg *et al* 2008, Kurz *et al* 2008). Natural resource development activities such as oil and gas well exploration and production and logging introduce additional complexity to disturbance regimes (Gauthier *et al* 2015, Shaw *et al* 2021) in various parts of the region (Williams *et al* 2021, 2013, Raynolds *et al* 2014, Pasher *et al* 2013).

Fire is a key driver of the carbon balance of boreal ecosystems (Harden *et al* 2000, Bond-Lamberty *et al* 2007, Wang *et al* 2021), but the relative importance and impacts of other disturbance types have been less studied (Shaw *et al* 2021). Thus, it remains unclear how much these other disturbance types and their interactions (Buma 2015) impact ABZ ecosystems. In this review, we summarize the existing state of knowledge of major disturbance types in North American ABZ ecosystems and use case studies of Landsat satellite-derived time series of vegetation greenness and moisture indices to illustrate the distinct spatiotemporal characteristics of vegetation loss and recovery associated with each disturbance type. Additionally, we review interactions between disturbances, which are likely to intensify in the future (Buma 2015, Seidl *et al* 2017).

In this review, we focus on “pulse” disturbances, characterized as generally abrupt, relatively discrete events that rapidly alter ecosystem structure, resources, or the physical environment (Pickett and White 1985). We do not address “press” disturbances which impact ecosystems slowly over decades and centuries (e.g., long-term warming; Grosse *et al* 2011). We divide major ABZ disturbances into several categories: 1) fire; 2) insects and pathogens; 3) permafrost-related disturbances; 4) anthropogenic disturbances; 5) weather-related disturbances; 6) riverine processes; and 7) ungulate and grazer activity. These disturbance types are not meant to be an exhaustive list of all known disturbances within the North American ABZ, but rather a characterization and discussion of the major climate-sensitive and anthropogenic disturbances within the region that impact vegetation processes. We do not, for example, include coastal erosion, alpine landscapes (e.g., avalanches), or localized geologic settings (e.g., volcanism).

By considering a range of major disturbance types, we seek to answer a set of interrelated questions: *What are the distinct causes of each disturbance type, and how are disturbance regimes (i.e., extent, frequency, and severity) sensitive to climate change and human activity? How does each disturbance type impact vegetation composition, structure, and recovery? How do different disturbance regimes interact with each other?* In doing so, we aim to provide context, identify data and knowledge gaps, and lay the groundwork for future studies that analyze how the full suite of disturbance agents are reshaping the vegetation dynamics of ABZ ecosystems.

2. Methods

This paper discusses the background, outstanding science questions, and data relevant to each of the seven broad disturbance categories. We also introduce case studies showcasing typical vegetation loss and recovery in response to select disturbances evident from remote sensing data.

2.1 Literature survey

Articles referenced in the background (Section 3), spatiotemporal characteristics (Section 4), and interactions (Section 5) sections were selected based on a thematic literature review as well as our own bibliographic lists derived from our active research in these fields. We searched the peer-reviewed literature using terms related to each disturbance category and type and biome (e.g., ‘boreal forest windthrow’, ‘cryoturbation’, ‘ice-jam flooding’). We emphasized recent (since 2014) papers and studies published on the North American boreal and Arctic ecosystems; however, we included studies from Eurasia to supplement topics where North American studies are lacking and to expand the global relevancy of this review.

2.2 Case studies and datasets

To evaluate patterns of vegetation loss and recovery after different disturbance types we compiled a set of locations (n = 397) of known disturbances within the North American ABZ to serve as case studies (Fig. 2). We compiled locations of known disturbance occurrences based on expert knowledge and field work of the authors as well as published locations in the literature and existing disturbance databases (Table S1). For each case study, we analyzed vegetation greenness and moisture changes during and following disturbance using time series of surface reflectance data from the Landsat series of satellites (1985-2020; Wulder et al 2019).

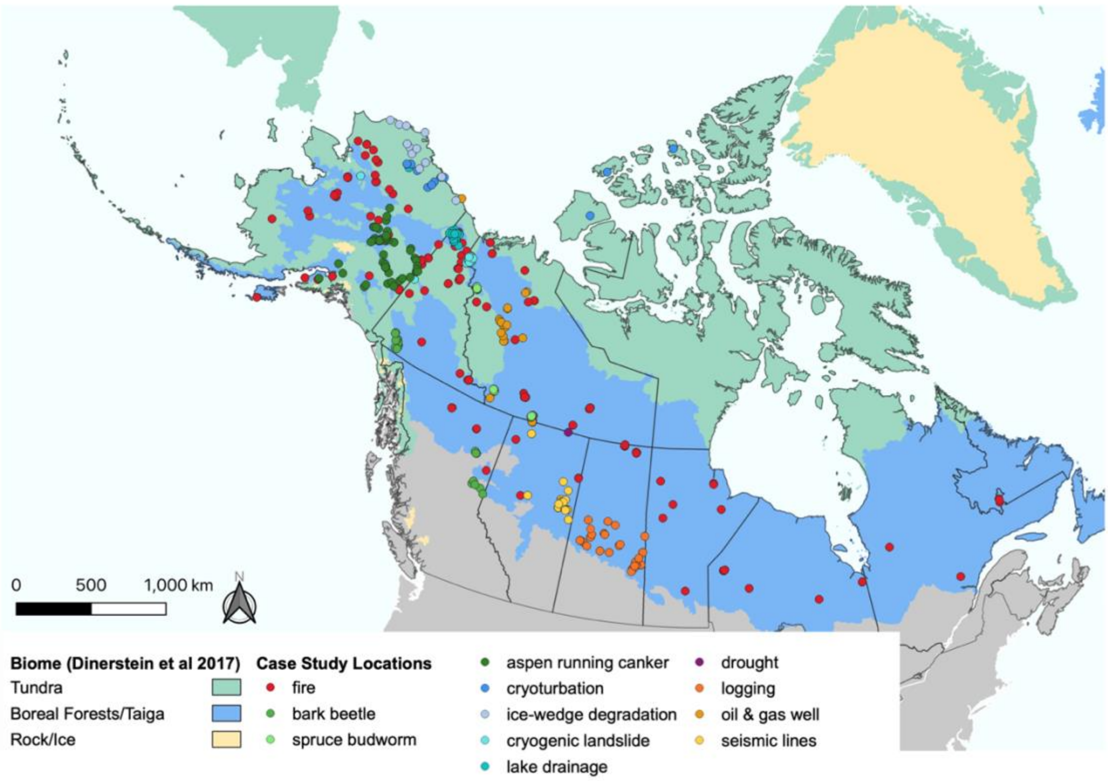

Figure 2. Locations of case study sites for disturbance types in the North American ABZ. Locations for spruce budworm and extreme drought provided by, and is the property of, the Forest Management Division, Department of Environment and Natural Resources, Government of the Northwest Territories.

For case study locations derived from individual latitude and longitude points, we extracted Landsat time series within a 100-m buffer surrounding each site to mitigate issues with geospatial accuracy of the case study locations. For case study locations derived from polygons, polygons were first filtered to only include “severe” impacts (if known), as well as disturbances that occurred between 2001 and 2016 to ensure adequate temporal coverage of pre- and post-disturbance vegetation greenness and wetness. The selected polygons were then randomly sampled (n = 25 per disturbance type), and 30 m Landsat pixels were randomly selected within each sampled polygon (n = 50 per polygon). For fire disturbance, in order to ensure broad coverage of diverse ecological conditions present within the North American ABZ, ten random points were sampled within each of five random fire polygons per Level II Ecoregion (US EPA 2015).

2.3 Case study analysis

We calculated spectral indices representing land surface greenness (the Normalized Difference Vegetation Index - NDVI; Rouse et al 1974, Tucker 1979) and wetness (the Normalized Difference Moisture Index - NDMI; Gao 1996). NDVI is a widely used index that is sensitive to leaf chlorophyll content and is generally correlated with vegetative cover and photosynthetic productivity. However, NDVI is less sensitive to changes in the state of evergreen forests (Jin et al 2017), which are the dominant forest type in the ABZ (Gauthier *et al* 2015). NDMI is an index that is sensitive to leaf water content and may reflect more subtle changes in vegetative stress in evergreen trees (Goulden and Bales 2019). While more specific and fine-scale indices may lend more information about, for example, species composition changes following disturbance, the use of NDVI and NDMI allows for broad coverage of the impact of different disturbances on vegetative cover and condition. Changes in NDVI and NDMI thus are interpreted as vegetation loss (e.g., declining NDVI or NDMI) and recovery (e.g., increasing NDVI or NDMI) in response to disturbance.

We developed time series of annual summer maximum greenness and wetness for the case study sites (Table S1). For each sampled location, we extracted all available Landsat 5, 7, and 8 surface reflectance data acquired each summer (day-of-year 151-242; May 31 –

August 31) from 1985 to 2020 for a total of ~11,000,000 multi-band measurements tallied across all pixels. These data were retrieved from the Landsat Collection 2 surface reflectance dataset (USGS 2021, Masek et al 2006), accessed using Google Earth Engine (GEE; Gorelick et al 2017) and functions provided by the *lsatTS* package (Berner et al 2021, Berner et al in review) in R (R Core Team 2021). We quality-screened these surface reflectance measurements based on pixel- and scene-criteria (i.e., scene-wide cloud cover < 80%, geometric uncertainty < 30 m, and solar zenith angle < 60 degrees) and further cross-calibrated them among Landsat sensors using the *lsatTS* package. Cross-sensor calibration is necessary to avoid spurious trends in NDVI and other spectral indices that arise from systematic differences in spectral bands among Landsat sensors (Sulla-Menashe et al 2016, Berner et al 2021). We calculated annual summer maximum surface greenness (NDVI) and wetness (NDMI) at each sampled location as the maximum summer NDVI or NDMI. Overall, we developed 14,709 annual time series of surface greenness and wetness for recently disturbed pixels across the study domain (Table S1).

Because some case study locations were approximate or derived from large aerial survey polygons, not all pixels were located over an actual disturbed area. Therefore, to focus our analyses on pixels that captured disturbance events, we filtered pixels to those that included detectable disturbance impacts on NDVI and NDMI within five years of the known disturbance event, except for cryoturbation and ice-wedge degradation, which occur within landscape mosaics and do not correspond to a single “event”. Aside from cryoturbation and ice-wedge degradation, disturbances were identified using visual interpretation of each time series and via the Breaks For Additive Season and Trend (BFAST) algorithm in the *bfast* package (Verbesselt and Herold 2012) in R (Fig. S1). BFAST iteratively estimates abrupt changes (or “breakpoints”) within time series and can be used to analyze seasonal and annual time series of satellite-observed reflectance to detect statistically significant temporal changes (Verbesselt and Herold 2012, Verbesselt et al 2010).

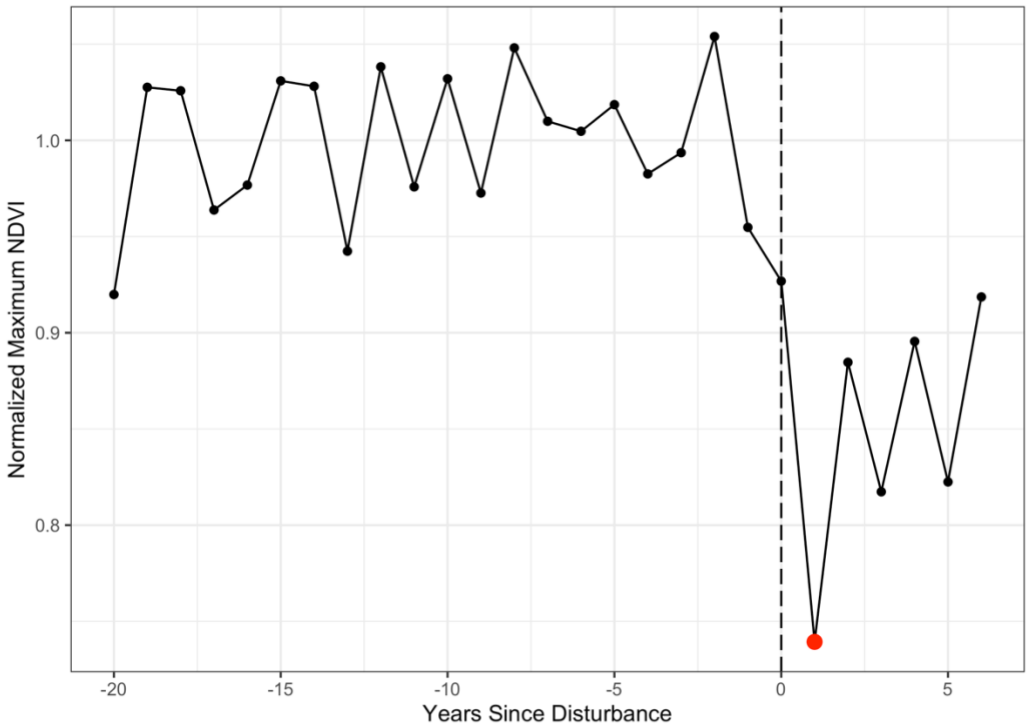

Figure 3. Example normalized maximum growing season NDVI for a site in the Northwest Territories, Canada (67.023º, -123.348º), where the NDVI is normalized to the pre-disturbance mean. A fire occurred in 2014 (dashed line). The red dot corresponds to the year of maximum impact of the fire on NDVI.

Following breakpoint detection, each time series with detected breakpoints was smoothed using the R function *smooth* **(**Tukey 1977)**,** and inflection points were identified in the smoothed time series. The series was first smoothed to identify “true” changes in the vegetation index trajectory, rather than those simply due to noise or interannual variability. The inflection point with the minimum (or maximum, for NDVI of lake drainage) spectral index value was identified as the year of full effect from the disturbance on land surface greenness and wetness. The time series before the breakpoint and following any breakpoints detected earlier in the series (e.g., between 1994 and 2014 in Fig. S1) was used to calculate an average pre-disturbance mean NDVI and NDMI. Each time series was then normalized by its pre-disturbance mean ($NDVI_{norm} = NDVI/NDVI_{mean}$, Fig. 3). We normalized the time series to better compare within and between disturbances, which occurred in different biomes and bioclimatic regions.

These normalized time series were used as our case study trajectories to evaluate the impact of each disturbance on vegetation as well as the magnitude, direction, and speed of recovery following each disturbance (see Section 3).

2.4. Disturbance characteristics and interactions

The major ABZ disturbance types have distinct spatial, temporal, and severity characteristics. To compare the spatial and temporal dynamics among disturbances, we developed several spatiotemporal metrics. *Spatial grain* describes the average extent of an individual disturbance event (e.g., for a wildfire it would be the size of a polygon associated with the outer perimeter of the burn scar, but for insect infestation it might be a single tree or forest stand). *Return interval* refers to the average length of time for the disturbance to reoccur in the same location. *Occurrence timeline* describes the average length of time a disturbance event lasts from initiation to completion (e.g., for wildfire: from ignition to extinction). *Recovery timeline* refers to the average length of time it takes for the vegetation/ecosystem to return to pre-disturbance conditions. Finally, *intensity/impact* refers to the average effect on vegetation and the ecosystem, from vegetation stress to complete vegetation mortality. We determined qualitative values for each of these categories and disturbance types using scientific literature and expert knowledge (see Section 4). The metrics were converted into relative numerical scales (Table S2) and applied to a principal component analysis (PCA) to understand how the different metrics correlate with one another across the different disturbance types. The PCA was conducted using the R function *prcomp*, with the categorical metrics scaled and centered within the PCA.

The degree to which different disturbance types interact with each other is complex and not well understood, and critical feedbacks between disturbances make their potential impacts difficult to analyze and predict. Therefore, we developed a disturbance interaction matrix based on our literature survey and expert knowledge. This matrix describes the impact (strong/weak positive, strong/weak negative, both, none, or unknown) of a "driver" disturbance on potential subsequent "response" disturbances (see Section 5). We distinguish "strong" and "weak" interactions by their relative effect on ecosystem structure and function, the ubiquity and likelihood of this impact occurring, and the ability of the ecosystem to resist or recover from subsequent response disturbances. For example, we classify the impact of boreal windthrow on subsequent insect and pathogen disturbance as "strong positive" (Fig. 20), because this interaction is a well-documented and impactful phenomenon within forested ecosystems (e.g., Malmstrom and Raffa 2000). In contrast, we classify the impact of logging on subsequent windthrow events as "weak positive" (Fig. 20), because while forest fragmentation, such as that created by forest harvest, does impart higher susceptibility to windthrow (Peterson 2004, Meilby *et al* 2001), the low probability of windthrow in boreal North America (Bouchard *et al* 2009) reduces the overall impact of this interaction. See Section 5 for a further discussion of these interactions.

3. Disturbance agents in North American Arctic and boreal ecosystems

3.1 Fire

3.1.1 Background

Wildfire is the most well-studied disturbance agent in forests of boreal North America, as fires have substantial impacts on human settlements (Kent 2017), subsistence resources (Nelson *et al* 2008), and air quality (Trainor *et al* 2009), in addition to climate (Randerson *et al* 2006, Potter *et al* 2020) and vegetation (Rogers *et al* 2013, Foster *et al*

2022). Fires in boreal North America are generally high-intensity crown fires that kill most affected trees and consume substantial belowground carbon stocks, in contrast to those in boreal Eurasia or more temperate ecosystems which include a high fraction of lower-severity surface fires that result in relatively low tree mortality (Stocks and Kaufmann 1997, de Groot *et al* 2013, Rogers *et al* 2015). Fire is less common in Arctic tundra but has been increasing in frequency and severity (Hu *et al* 2015, McCarty *et al* 2021), especially in the Beringian region (Rocha *et al* 2012, Gaglioti *et al* 2021, Racine *et al* 1985, Masrur *et al* 2018). Recent increases in boreal and Arctic wildfire activity may indicate fundamental shifts in the causes and impacts of the underlying fire regime, including overwintering fires that smolder during winter months and reappear the following year (Scholten *et al* 2021, Xu *et al* 2022), increased occurrence of lightning ignitions (Veraverbeke *et al* 2017, Chen *et al* 2021c), and long-term shifts in forest composition following these fires (Baltzer *et al* 2021, Mack *et al* 2021). Forest fire records throughout the North American boreal region show an increase in annual burned area and number of large fires since the mid-20th century (Hanes *et al* 2019, Calef *et al* 2015, Walker *et al* 2020b). The majority of projections of future fire regimes suggest increasing fire activity across boreal North America over the 21st century due to climate change (Bachelet *et al* 2005, Amiro *et al* 2009, Hope *et al* 2016, Veraverbeke *et al* 2017, Chen *et al* 2016, Wang *et al* 2020, Phillips *et al* 2022).

Precursors to fire in boreal ecosystems are well understood - an adequate amount of fuel and fuel dryness are required for fires to ignite and spread, in addition to ignition sources such as lightning strikes and anthropogenic activities (Veraverbeke *et al* 2017, Archibald *et al* 2018, Rogers *et al* 2020). In the boreal zone, fires are generally limited by fuel dryness and ignition sources because the characteristically deep organic and moss layers provide ample fuel. Both species composition and litter moisture are influenced by site drainage conditions, with organic-rich soils dominated by fire-prone and flammable species such as black spruce

(*Picea mariana*). Conversely, Jack pine (*Pinus banksiana*) and less flammable deciduous species typically occur in well-drained locations with thinner, drier soils (Walker *et al* 2018, 2020b).

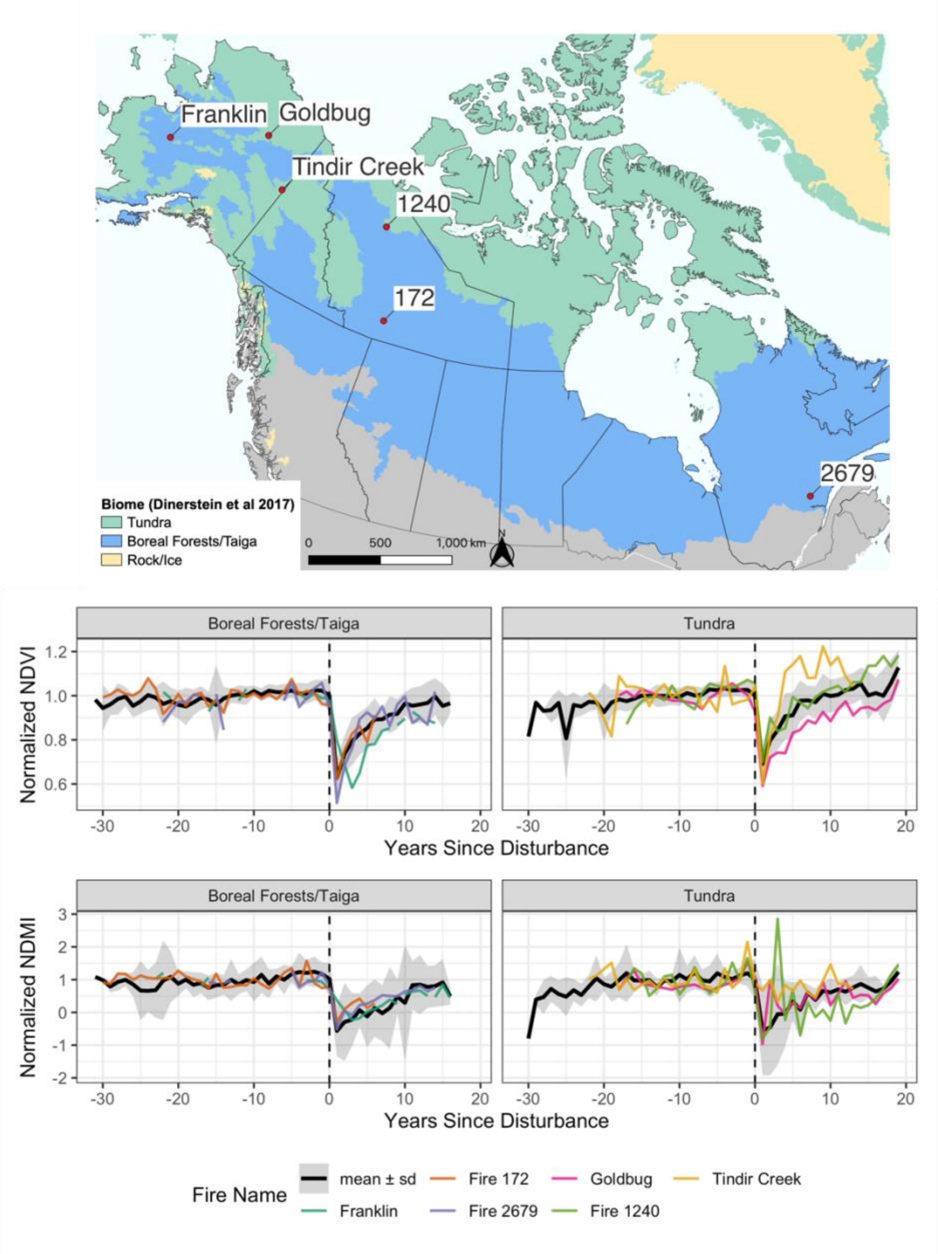

Figure 4. Average (n = 32) as well as six individual case study trajectories for fire disturbances in Alaska and Canada showing NDVI and NDMI normalized to the pre-disturbance average value.

Lightning strikes ignite most fires in the North American ABZ. Lightning ignitions have increased since the mid-20th century due to a warmer and more convective atmosphere (Veraverbeke *et al* 2017, Chen *et al* 2021c). More severe fire weather is also prolonging fire seasons and increasing fire intensity and annual area burned. For example, Kasischke *et al* (2010) found the mean annual of area burned in Alaska during the 2000s was 50% greater than any previous decade since the start of the record in 1940, resulting in increased ground-layer combustion and net carbon emissions to the atmosphere (Turetsky *et al* 2011).

Within the North American boreal region, fires create lasting legacies on vegetation, driving changes in soil characteristics, regeneration patterns, and successional trajectories (Johnstone *et al* 2010, Gaglioti *et al* 2021, Mack *et al* 2021). High-severity forest fires that remove much of the organic layer favor regeneration by deciduous and fast-growing pine species, which may maintain dominance under a warming climate (Johnstone *et al* 2011). Field data have also suggested that increased warming and fires may be altering the ability of typically resilient black spruce forests to recover following large fires, leading potentially to a tipping point for boreal vegetation – shifting from evergreen to deciduous or non-forested land cover types (Baltzer *et al* 2021). Alterations to phenological metrics from time series of NDVI and other greenness metrics observed in burned areas in Alaska may also indicate long-term shifts in vegetation cover type and photosynthetic activity at regional scales (Potter 2020, Madani *et al* 2021).

In the Arctic tundra, our understanding of the drivers of the wildfire regimes is less thorough, due to a combination of factors including lower fire frequency, remoteness, and limited in-situ observations. It is commonly believed that lightning (He *et al* 2022, Chen *et al* 2021c), summer temperature, and precipitation (Hu *et al* 2015, Vachula *et al* 2022) are among the primary factors controlling the wildfire regimes in Alaskan tundra. Fire usually favors the recruitment and growth of deciduous shrubs in the tundra. It is therefore an important mechanism for Arctic shrubification (Lantz *et al* 2010b, Jones *et al* 2013, Frost *et al* 2020). Following fire, net ecosystem productivity (NEP) declines because of reduced vegetation productivity and increased ecosystem respiration, with forest ecosystems becoming a carbon source for roughly one to two decades (Amiro *et al* 2010, Kurz *et al* 2013, Liu *et al* 2011). In the tundra, vegetation productivity recovers more quickly, in as little as three years post-fire, though longer term impacts on NEP remain less clear (Gaglioti *et al* 2021). As vegetation and soils recover, NEP increases up to a maximum and then decreases to a steady state, at which point the ecosystem is again carbon neutral or a carbon sink (Goetz *et al* 2012, Song *et al* 2018). Climate change, however, may alter the post-fire NEP response in the future due to species composition shifts, productivity changes, and permafrost thaw (Rocha *et al* 2012, Foster *et al* 2019, Mekonnen *et al* 2019, Baltzer *et al* 2021, Gibson *et al* 2018).

Vegetation responses to fire disturbance can be seen in Landsat-derived trajectories of greenness (NDVI) and wetness (NDMI), as showcased in the average across all fire trajectories (n = 32) as well six individual fires (Fig. 4). The average trajectory shows a rapid decline in normalized NDVI and NDMI immediately following fire, with a moderate recovery rate in the following years (approximately 10 years for NDVI and 15 years for NDMI). Tundra NDVI recovers more rapidly, with NDVI values reaching the pre-disturbance mean within a decade following fire. The NDMI response following fire is more varied for the tundra locations, a pattern which highlights the cascading effects of wildfire on accelerated permafrost thaw and associated changes in soil thermal and moisture regimes, and variability arising from local differences in fire severity and ground ice conditions (Jones *et al* 2015).

3.1.2 Limitations, data needs, and unknowns

Large fire databases are crucial for understanding fire precursors, effects, trends, and dynamics in boreal and Arctic ecosystems. In Alaska and Canada, existing fire history databases provide fire perimeter polygons beginning in the 1940s and 1960s, respectively, and are maintained and updated annually. These databases are some of the longest and most complete large-scale historical fire records available anywhere on the planet (Kasischke *et al* 2002, Stocks *et al* 2002) and they are foundational datasets for investigating regional impacts of post-fire vegetation succession (Rogers *et al* 2013, Potter *et al* 2020). Despite this, due to the great challenges in mapping wildfires in the high latitudes (e.g., limited availability of Landsat observations during a short growing season and persistent cloud cover; Chen *et al* 2021b, 2021a), omissions of large wildfire events by these wildfire history records still exist, particularly in the tundra (Jones *et al* 2013). Moreover, the fire perimeters themselves become less accurate further back in time, and often contain substantial patches of unburned vegetation (Kasischke *et al* 2002, Potter *et al* 2020, Walker *et al* 2018). Advances in remote sensing tools enable fires and their impacts to be mapped and tracked at increasingly finer spatiotemporal resolutions (Duncan *et al* 2020, Hall *et al* 2020, Eidenshink *et al* 2007). Field data are also crucial for studying fire impacts on carbon stocks and fluxes, vegetation recovery, hydrology, and other ecosystem properties, and a growing number of databases are allowing for meta-analyses of fire impacts (Walker *et al* 2020a, Virkkala *et al* 2022, 2018). However, additional combustion estimates are needed to better understand the interactions between fire weather, fire spread and intensity, and combustion (Walker *et al* 2020b).

Further data are required to elucidate the interactions between wildfire, vegetation, and permafrost in the context of changing climate (Treharne *et al* 2022, Gibson *et al* 2018). Increasing temperatures, changing precipitation, and increases in fire activity will impact vegetation composition and structure, hydrology, and carbon fluxes. Future researchers could utilize a combination of active radar and subsidence data, high spatial and spectral resolution imagery, digital elevation models (DEMs), and airborne LiDAR and other remote sensing data to observe and analyze these changes. It is also unclear how these changes to vegetation and fuels will interact with future fire regimes. Predicted increases in deciduous fraction and declines in organic layer and other fuels (Foster *et al* 2019, Mekonnen *et al* 2019) may lead to decreasing fire frequency and severity, even as fire weather and fuel drying increases (Parks *et al* 2015). Further, if young stands re-burn following fire, it is unknown how and which species may be able to regenerate as seed banks become depleted and soils become less conducive to seedling establishment (Baltzer *et al* 2021).

From a societal perspective, the increasing frequency of large fires, and necessary increased investments in fire-fighting activities at the wildland-urban interface, will strain the existing fire management budgets and governance structures (Rogers *et al* 2020). More studies are needed linking the influence of management on fire regimes, both historically and in the future, to quantify these relationships and make predictions for the efficacy and costs of fire management efforts (Melvin *et al* 2017b, Calef *et al* 2015, Phillips *et al* 2022).

3.2 Insect outbreaks and pathogens

3.2.1 Background

Biotic disturbances, such as fungal pathogens (e.g., root rots and needle rusts) and insect outbreaks (e.g., bark beetles and defoliators/leaf miners) can cause extensive tree mortality during outbreaks (Holsten *et al* 2008, Kautz *et al* 2016). Fungal pathogens often kill individuals slowly by disrupting water and nutrient transport (Holsten *et al* 1985) and reducing growth. In contrast, episodic insect outbreaks can cause major growth reductions and spatially widespread tree mortality over a few years, at times eclipsing that due to fire. For example, annual forest volume lost due to productivity reduction and mortality from pests and pathogens in Canada was estimated to be 106 million m^3 per year between 1982 and 1987, which was three times that lost annually to fire and 70% of volume harvested in Canada nationally during that period (Hall and Moody 1994, Malmstrom and Raffa 2000, Volney and Fleming 2000, Price *et al* 2013). In the 1990s in Alaska, insects cumulatively damaged 1.6-2 million hectares of forest, which was 30% more area than burned during that period (Malmstrom and Raffa 2000).

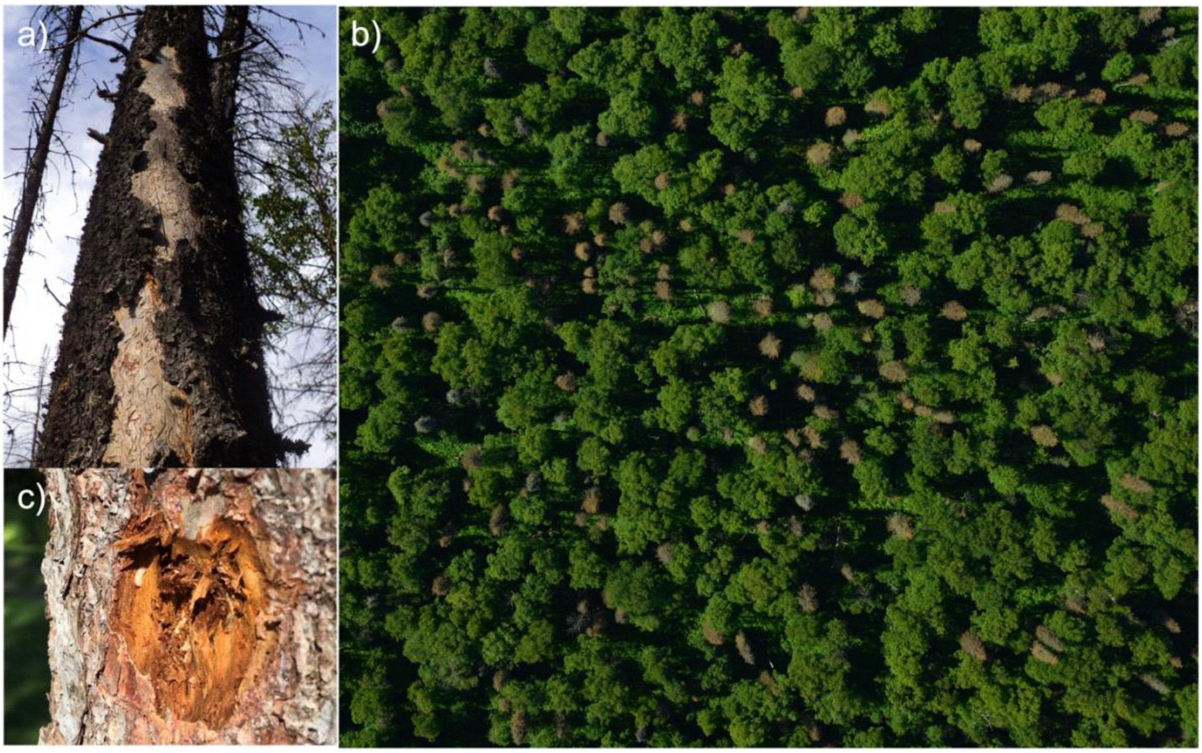

Figure 5. a) Tree trunk infested with mountain pine beetle, showcasing egg galleries; b) aerial imagery of white spruce infested with spruce beetle, south-central Alaska, credit GLiHT; c) spruce beetle larvae within a white spruce trunk

Bark beetles, such as the mountain pine beetle (*Dendroctonus ponderosae*) and spruce beetle (*Dendroctonus rufipennis*), kill host trees outright by feeding on the cambium and phloem (Fig. 5a,c) and disrupting water transport (Malmstrom and Raffa 2000, Bentz *et al* 2010). These beetles attack trees through "mass attacks" of many beetles, attracted via massing pheromones released by the beetles (Raffa *et al* 2008). Bark beetle populations typically exist at relatively low levels, punctuated by occurrences of high, epidemic levels due to climate-, disturbance-, or forest structure-related triggers (DeRose *et al* 2013, Seidl *et al* 2016). Young, healthy trees can often defend against low levels of attacking beetles by exuding resin and allelochemicals. However, stressed trees and those experiencing a large number of attacking beetles are more likely to succumb to infestation (DeRose and Long 2012). Thus, conditions that lead to vegetation stress, such as drought, often lead to outbreak events (Sherriff *et al* 2011, Seidl *et al* 2016).

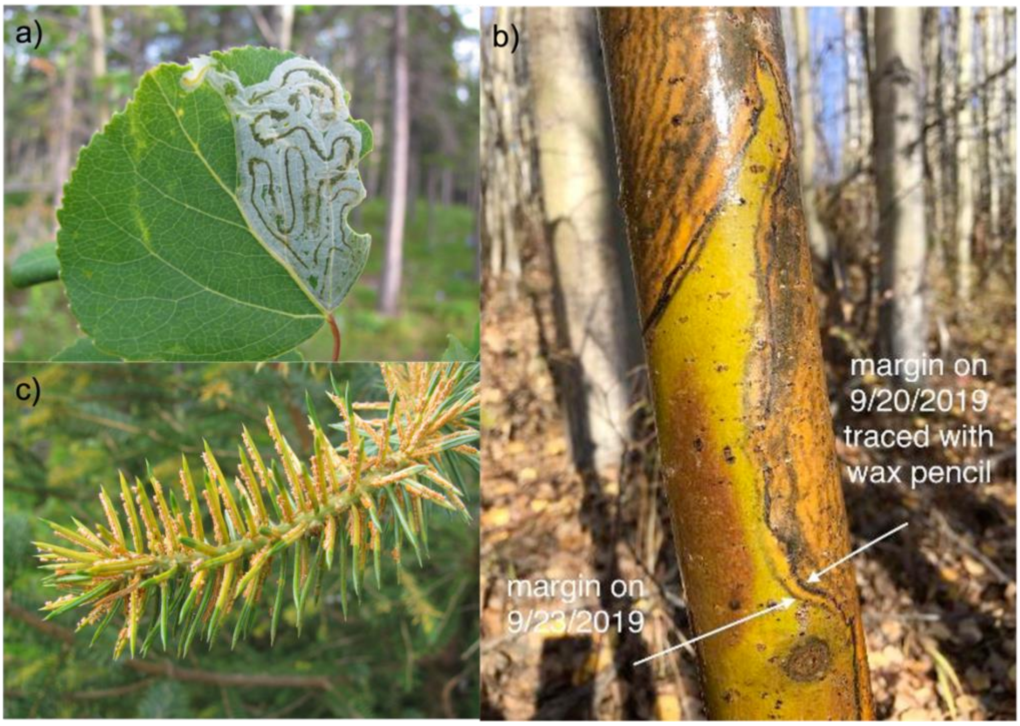

Figure 6. a) Mines and larvae of an aspen leaf miner, USDA Forest Service photo by Robin Mulvey; b) advance of an aspen running canker over the course of just three days in 2019, USDA Forest Service photo by Lori Winton; c) spruce needle rust on a Sitka spruce, USDA Forest Service photo by Robin Mulvey. Photos from the USDA Forest Service public Flickr Page (https://www.flickr.com/people/194703066@N07/).

Defoliators and leaf miners feed on the leaves and needles of host plants. In the North American ABZ, these guilds include, for instance, eastern and western spruce budworms (*Choristoneura* spp.), Jack pine budworm (*Choristoneura pinus*), aspen leaf miner (*Phyllocnistis populliella*) (Fig. 6a), and large aspen tortrix (*Choristoneura conflictana*). Outbreaks of these defoliators and miners cause significant tree growth reduction and potentially tree mortality. Removal or damage to needles and leaves disrupts water transport and interferes with photosynthesis, which can kill trees directly or cause physiological stress that predisposes them to death from other factors, such as drought (Malmstrom and Raffa 2000). Recovery from major defoliation and mining depends on the extent of damage and the amount of carbon reserves held in other tissues (Boyd *et al* 2021). Deciduous species generally are more able to re-foliate from leaf damage than evergreen species, even in the same year as defoliation (Krause and Raffa 1996, Holsten *et al* 2008). Evergreen species, however, often have a high rate of mortality following successive years of intense defoliation, potentially leading to species composition shifts post-outbreak.

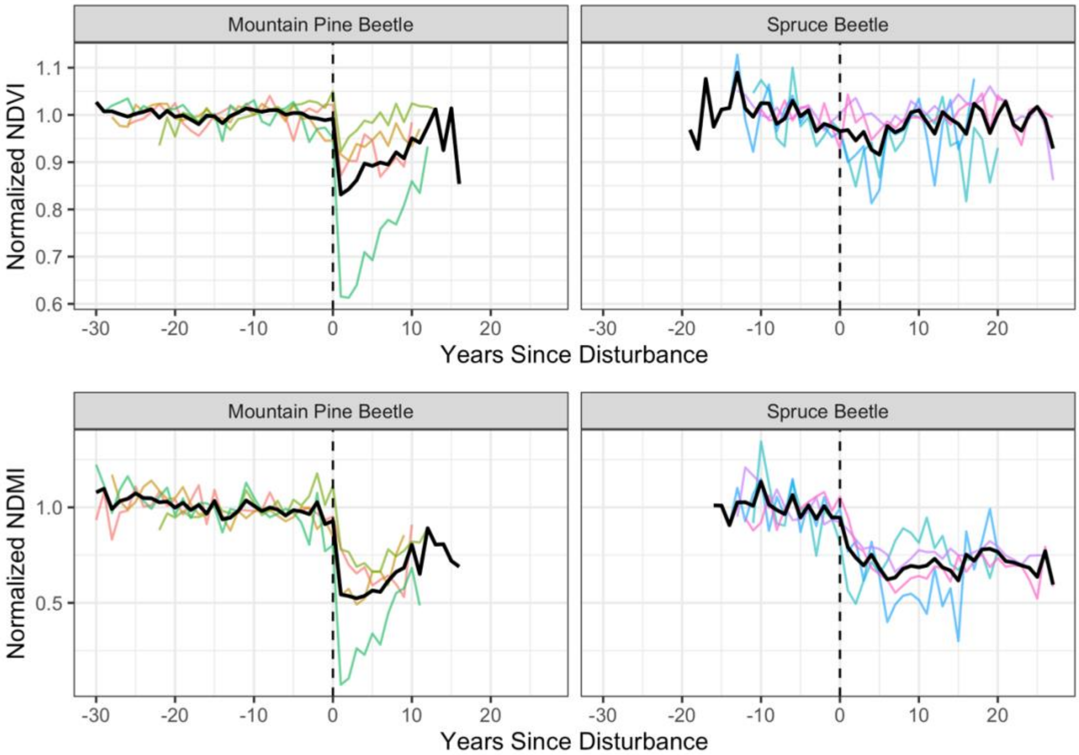

Figure 7. Average (black) and individual (colors) case study trajectories for mountain pine beetle (British Columbia; n = 4) and spruce beetle (Yukon Territory; n = 4) outbreaks showing NDVI and NDMI normalized to the pre-disturbance average value.

The most common pathogens in the North American ABZ include root rot (e.g. *Inonotus tomentosus*), heart rot fungi (e.g. *Fomitopsis pinicola*), and needle rusts (e.g. *Chrysomyxa ledicola*; Fig. 6c) (Armstrong and Ives 1995, Holsten *et al* 2008). These pathogens can cause hydraulic impairment by damaging vascular systems, reduce productivity through impacts on needles and leaves, and ultimately lead to plant mortality. Recently, an outbreak of the novel aspen running canker (*Neodothiopora populina*) caused widespread mortality of quaking aspen (*Populus tremuloides*) in interior Alaska (Fig. 6b). Aspen mortality from these infections was exacerbated by ongoing drought as well as an outbreak of aspen leaf miner (Ruess *et al* 2021).

While pathogens frequently affect a wide range of species, insects are often species- or genus-specific in their host requirements (Armstrong and Ives 1995, Holsten *et al* 2008). Hosts that are larger, older, or stressed are generally more susceptible to bark beetles. Thus, areas with high numbers of susceptible hosts are most vulnerable to insect outbreak, with mature, host-dominated stands being the most susceptible (Raffa *et al* 2008, Chapman *et al* 2012, DeRose *et al* 2013, Hart *et al* 2015). These homogenous stands provide a high quality habitat for insects, allowing for self-sustaining populations and sources of large-scale outbreaks (Malmstrom and Raffa 2000, Seidl *et al* 2016). The relatively low biodiversity in ABZ forests thus makes them particularly vulnerable to insect and pathogen outbreaks (Senf *et al* 2017a, Campbell *et al* 2008, Jactel *et al* 2005). Increasing temperatures and drought are thus generally expected to increase the impacts of insects and pathogens in the North American ABZ.

Insect and pathogen outbreak dynamics are affected and compounded by climate and weather by influencing the range and population size of insects and pathogens and altering the vulnerability of plants. For example, warming temperatures can reduce wintertime mortality and accelerate population growth of insects like the spruce beetle (Raffa *et al* 2008, Bentz *et al* 2010, Gray *et al* 2013). Spruce beetles usually have a two-year (semivoltine) life cycle, but warmer conditions can accelerate larval growth, causing a shift to a one-year (univoltine) life cycle (Hansen *et al* 2011). More beetles with univoltine life cycles drives faster population growth and more severe outbreaks, such as occurred with the expansion of bark beetle outbreaks in British Columbia in the 1970s and 1980s (Bentz *et al* 2010). Host plants also interact with climate through host stress levels. Drought predisposes trees to disease and infestation (Raffa *et al* 2008, McKenzie *et al* 2009, Boyd *et al* 2021, Ruess *et al* 2021), and can be a secondary cause of mortality following defoliation stress (Malmstrom and Raffa 2000). Climate change is predicted to result in range expansion of insect species (de la Giroday *et al* 2012) and increases in outbreak severity and frequency (Raffa *et al* 2008). In Alaska, drought, high vapor pressure deficit, and high temperatures are key contributors to mortality linked with aspen leaf miner and aspen canker (Ruess *et al* 2021, Boyd *et al* 2021).

Because bark beetles tend to affect one or only a few host tree species and preferentially attack larger trees, their outbreaks often result in a shift towards smaller size classes and non-host species (Veblen *et al* 1991, Campbell and Antos 2015, Zeppenfeld *et al* 2015). Productivity often increases in these subsequent stands as non-infested trees are released from suppression (Campbell *et al* 2019). In more homogenous stands, species composition can shift towards early successional species after an outbreak. These impacts can be seen in trajectories of NDVI and NDMI before and during outbreaks (Fig. 7). Defoliators also tend to impact one or a few species - the eastern spruce budworm (*C. fumiferana*) mostly infests balsam fir (*Abies balsamea*) and white spruce (*Picea glauca*), and infestation-caused mortality often leads to release of seedlings and saplings of host species (Boulanger and Arsenault 2004). Changes in NDVI are generally subtle as outbreaks build, sometimes asynchronously, within individual trees (Fig. 5b), and are usually only visible in moderate-resolution satellites when large areas are impacted (DeRose and Long 2012). This subtle NDVI pattern (Fig. 7) is especially characteristic of spruce beetle outbreaks, which do not exhibit a characteristic “red-stage” attack as do pine species infested with mountain pine beetle (Coops *et al* 2006). However, NDMI often does decline (Fig. 7), due to decreases in transpiration and increases in foliar water stress during and following bark beetle outbreaks (Foster *et al* 2017). In contrast, trajectories of NDVI and NDMI during and following spruce budworm infestation in the Northwest Territories (Fig. 8) have a clearer signal, with some variability across the individual sites, highlighting the impact of infestation severity on the spectral signal. Sites which have a lower infestation severity (e.g., percent defoliation) will have a more subtle signal than sites which had more complete defoliation from spruce budworm infestation (Senf *et al* 2016). The response of NDVI to aspen running canker is also clear, with limited recovery following the drop in NDVI due to infestation (Fig. 8). NDMI response is less clear, with some decline following infestation.

3.2.2 Limitations, data needs, and unknowns

Past insect outbreaks are often identified through dendrochronology and pollen records (Sherriff *et al* 2011, Anderson *et al* 2010). However, these are limited to specific locations, usually where an outbreak is known, resulting in biases in our understanding of their extent and occurrence. Aerial detection surveys that produce polygons of infestation extent and severity are valuable for determining the regional and national impacts of forest pests. However, these polygons are often at a coarse spatial scale with potentially low positional accuracy (Wulder *et al* 2006, Hall *et al* 2016). Detection of recent or ongoing outbreaks using moderate resolution satellite sensors is possible, especially for large, severe outbreaks (Hall *et al* 2016, Meddens and Hicke 2014, Senf *et al* 2017a, 2016). Specialized methods are generally required for each insect type (e.g., bark beetles vs. defoliators). Foliar color changes of conifers infested with bark beetles often progress from green, sometimes to red, and to gray as needles lose moisture and are ultimately shed from the tree. The red and gray stages are easily detectable in multispectral imagery (Coops *et al* 2006), however the green stage is more subtle, making early detection difficult (DeRose *et al* 2011). Despite this difficulty, some studies have had success in using the water-sensitive shortwave infrared wavelengths to detect early moisture stress from green-stage infestations (Foster *et al* 2017).

Accurate and temporally and spatially consistent datasets of infestation/infection status and extent across jurisdictions are crucial for determining the extent and severity of past and ongoing outbreaks, and for predicting future outbreaks (Kautz *et al* 2016, Senf *et al* 2017b). Such large-scale datasets would also aid in generalizing detection methods across

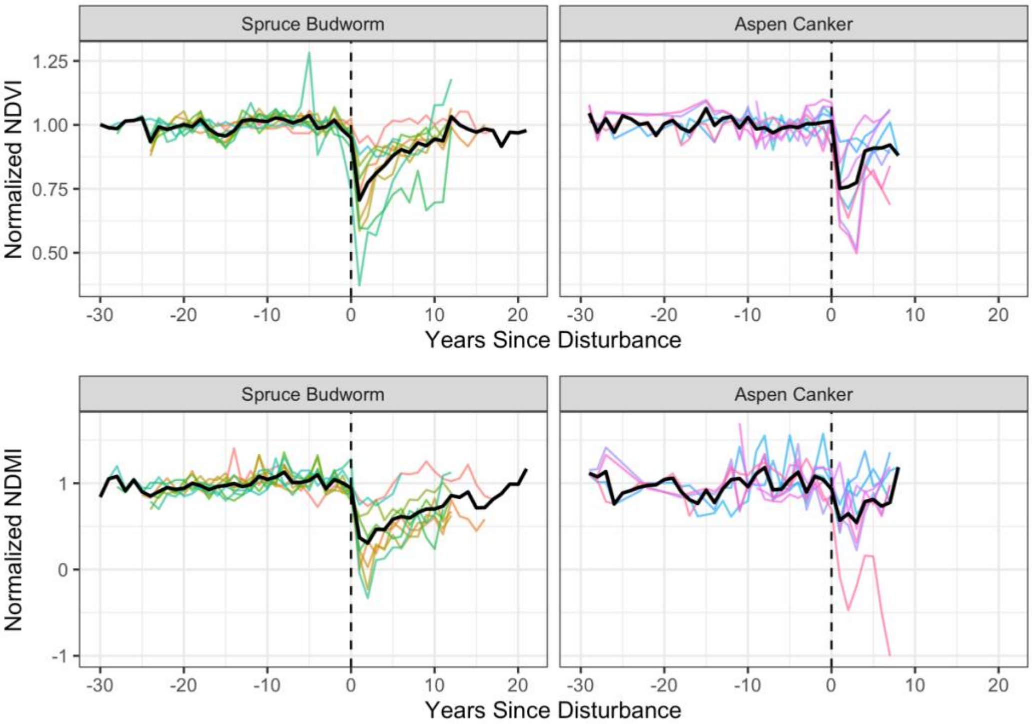

Figure 8. Average (black) and individual (colors) case study trajectories for spruce budworm infestation sites in the Northwest Territories (n = 9) and aspen running canker infestation sites in Alaska (n = 8) showing NDVI and NDMI normalized to the pre-disturbance average value.

wider regions and disturbance agents. Because some major limitations to detecting insect and pathogen disturbance from remotely sensed data include accurately discriminating between these disturbances and other vegetation stressors, due to the exhibition of similar spectral signals (Senf *et al* 2017b), field observations of infestation status that are coincident with remote sensing observations will assist in developing more accurate algorithms for multi-stage detection efforts (Cessna *et al* 2021). Increased availability of different types of remote sensing data, particularly hyperspectral and radar imagery, have the potential to identify changes in forest moisture related to insect and pathogen outbreaks at regional scales and with high spatial detail.

Studies have shown that insects and pathogens are expanding their ranges poleward with increasing temperatures, increasing the area of forest vulnerable to outbreak (de la Giroday *et al* 2012, Pureswaran *et al* 2018). Insects are also beginning to infest novel host species (NRC 2018), and it is unclear how such host species will respond. Such range expansion highlights the need for increased detection and monitoring of outbreaks, as well as the need for predictions of future infestation vulnerability.

3.3. Permafrost-related disturbances

Throughout much of the northern high-latitudes, ecosystems are underlain by permafrost, or soil that remains frozen for more than two years (Gruber 2012). However, with climate change, permafrost ground temperatures are increasing (Biskaborn *et al* 2019) and the active layer – the upper layer of soil that thaws in the summer – is becoming deeper across large areas (Smith *et al* 2022). In addition to the active layer, the physical structure of these soils is being altered across many landscapes in the ABZ due to extensive changes to permafrost status due to warming, and permafrost thaw is expected to increase further in the future, both linearly and abruptly (Kokelj *et al* 2015, Turetsky *et al* 2020). These changes in physical structure can dramatically alter the topography, hydrology, and vegetation, resulting in heterogeneous topography and thermokarst features, especially in ice-rich locations. In this section, we describe several unique disturbances in the ABZ and their associated permafrost-related processes, including cryoturbation, ice-wedge degradation, cryogenic landslides, and lake drainage.

3.3.1. Cryoturbation

Permafrost soils often exhibit warped or broken soil horizons that result from cryoturbation, the frost-based movements of seasonally frozen materials (Bockheim and Tarnocai 1998). Cryoturbation can also create distinctive surficial disturbance features that generate fine-scale spatial heterogeneity in ground conditions and serve as foci for ecological change (Walker *et al* 2011, Frost *et al* 2013, Aalto *et al* 2017). Frost circles are a common form of patterned ground. They occur as approximately circular patches (~0.5-3 m diameter) of mineral soil that often form geometric mosaics of vegetated and unvegetated microsites at uniform spacing of ~1–3 m (Fig. 1c).

Frost circles are common in permafrost regions, particularly where surface organic material is lacking and the soil profile is dominated by fine-textured silt or clay (Bockheim *et al* 1998, Peterson and Krantz 2003). The intensity of cryoturbation is strongly affected by soil moisture, soil texture, changes in seasonal temperature, and snow cover (Aalto *et al* 2017, Daanen *et al* 2007). In general, climate warming and increased snow cover dampen cryoturbation by reducing differential frost-heave. Climate warming can also dampen cryoturbation indirectly by promoting vegetation colonization, which stabilizes the soil and results in organic matter accumulation on cryoturbated soils. Species that are fast-growing and/or tolerant of ground surface disturbances are best able to colonize cryoturbated surfaces (Kade *et al* 2005, Sutton *et al* 2006, Frost *et al* 2013). Once cryoturbation is reduced or no longer occurring, the increase in biomass is abrupt and persistent; however, cryoturbation can be renewed if vegetation and organic material are removed by other disturbances (chiefly wildfire).

Cryoturbation can have nonlinear responses to climate change with respect to vegetation cover and biomass, which can be detected in multi-decadal NDVI time series (Frost *et al* 2014). Furthermore, cryoturbation has distinctive spatiotemporal properties as a disturbance agent, because features usually occur as a multitude of 1-3 m microsites within a broad landscape mosaic, and the disturbance acts annually and is not episodic. At our case study locations, both NDVI and NDMI increased over the 30-year Landsat record (Fig. 9). With respect to NDVI, the warming climate could allow vegetation to colonize previously bare frost circles in cryoturbated landscapes, which would reduce further cryoturbation (Frost *et al* 2013). The NDVI increase could additionally reflect a general background greening (i.e., vegetation increase) of the landscape, as only a fraction is cryoturbated, and the remainder can have nearly complete vegetation cover. For NDMI, the increase in moisture is likely in part due to the moisture content of the colonizing vegetation, but also increased soil moisture beneath the vegetation cover (Fig. 9)

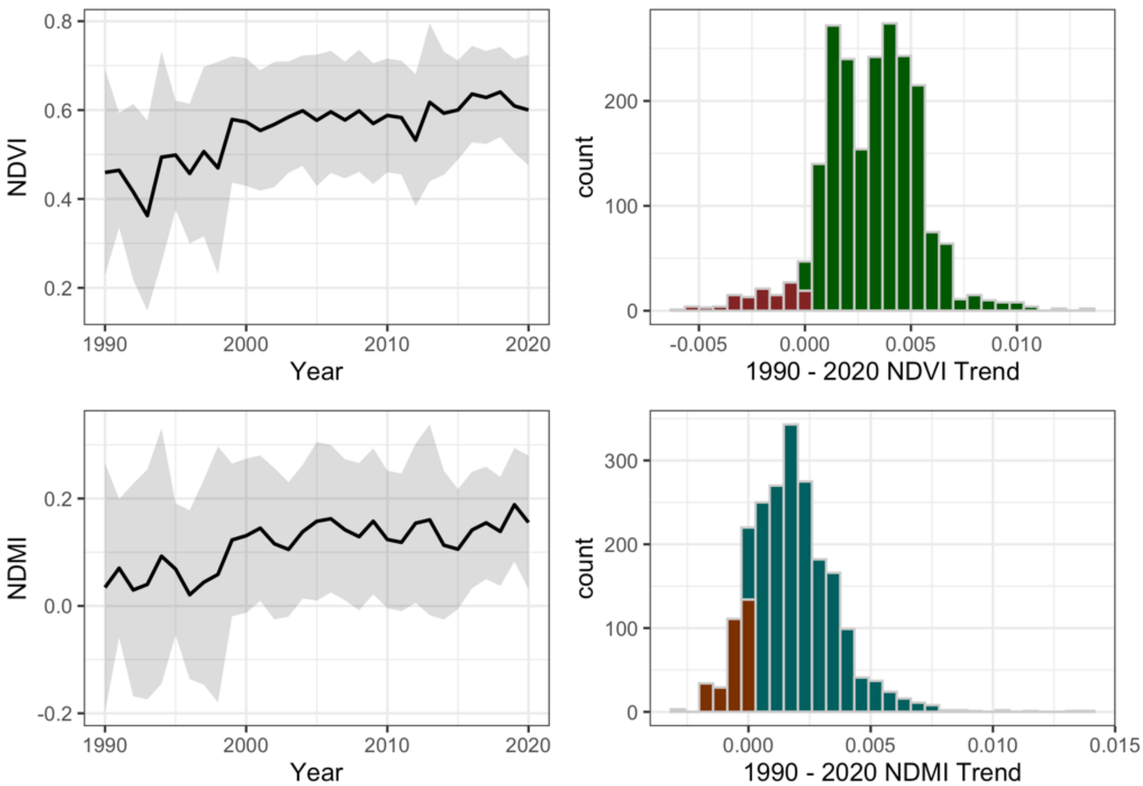

Figure 9. Average trajectories for NDVI and NDMI (left, not normalized) and distributions of trends (right) for NDVI and NDMI at cryoturbation case study pixels (n = 2129, across 47 sites) in northern Alaska. Trends were calculated as the slope from linear models fit at Landsat pixel for the vegetation index. Note that these trajectories were not normalized because cryoturbation is an ongoing disturbance, rather than a single event.

3.3.2. Ice-wedge degradation

Polygonal ground, encompassing mosaics of ice-wedge polygons (~5-15 m wide) formed by contraction cracking followed by annual cycles of thawing and refreezing, is widespread and conspicuous in permafrost landscapes (Liljedahl *et al* 2016). Wedge-shaped masses of ice underlie the edges of each polygon (Fig. 10). Ice-wedge degradation occurs when the uppermost portions of ice wedges thaw, which triggers local ground subsidence, ponding, and persistent changes to vegetation and hydrologic connectivity across the landscape (Fig. 10). Ice-wedge degradation often results in substantial micro-topographic changes, such as the transition from low-centered to high-centered polygonal landforms.

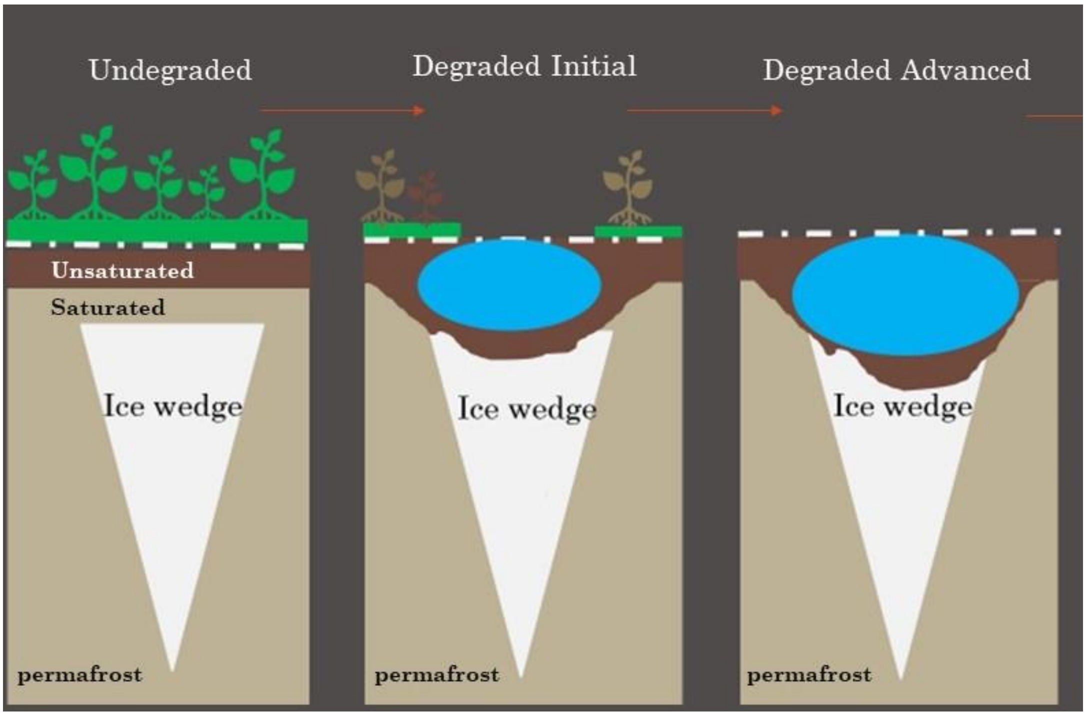

Figure 10. Schematic of ice-wedge degradation showing thawing of ice wedges and associated ponding and vegetation change. Image credit: Kelcy Kent.

Polygonal ground is most common in tundra with continuous permafrost, especially areas with fine-textured soils, and patterned landscapes can cover areas as large tens of square kilometers or larger (Lachenbruch 1962). However, ice wedges are also common in discontinuous permafrost regions well into the boreal forest (Swanson 2016, Kokelj *et al* 2014). Extreme warm and wet summers initiate ice-wedge degradation (Liljedahl *et al* 2016, Jorgenson *et al* 2006, 2015). Long periods of time (i.e., millennia) without additional disturbances are required to develop large ice wedges, so the terrain affected by ice-wedge degradation has historically supported “climax” vegetation communities – usually tussock tundra or needleleaf woodlands in boreal forest settings (Billings and Peterson 1980).

Local and regional variability in the timing and extent of ice-wedge degradation arises from differences in surficial materials and ground-ice content, disturbance history (natural and anthropogenic), regional climate gradients, and regional differences in the timing and magnitude of recent extreme warm summers (Raynolds *et al* 2014, Kanevskiy *et al* 2017, Frost *et al* 2018a, Farquharson *et al* 2019). This variability in ice-wedge degradation contributes to variability in patterns of tundra vegetation change (e.g., tundra greening or browning). Once thaw begins, the resultant subsidence forms small, flooded pits and troughs along the polygon margins. These pits and troughs pock-mark the landscape, kill existing vegetation that is adapted to mesic conditions (i.e., a mechanism for tundra browning) (Lara *et al* 2018), and support the colonization of hydrophytic vegetation (e.g., wetland sedges and mosses). Secondary impacts can affect large areas because the generation of pits and troughs creates new hydrologic flowpaths that alter soil hydrology and the distribution of surface water (Koch *et al* 2018). Over time (usually a matter of years to a decade), most pits and troughs become colonized by wetland vegetation, and surface water extent declines due to the development of an organic mat (i.e., a mechanism for tundra greening) (Wolter *et al* 2016).

Successional processes after ice-wedge degradation could explain in part the increasing NDVI trajectories in ice-wedge polygon landscapes (Fig. 11). However, this increase is likely also being driven by a general background greening of the tundra landscape in response to climate warming (Myers-Smith *et al* 2020, Berner *et al* 2020), as the affected microsites comprise only a fraction of the broader polygonal landscape. The distribution of NDVI dynamics includes numerous pixels with strong “browning” signals, probably due to extensive ice wedge degradation and increasing surface water (Jorgenson *et al* 2022). The increasing NDMI (Fig. 11) over time in these landscapes is likely being driven by the increasing surface water due to the development of pits and troughs.

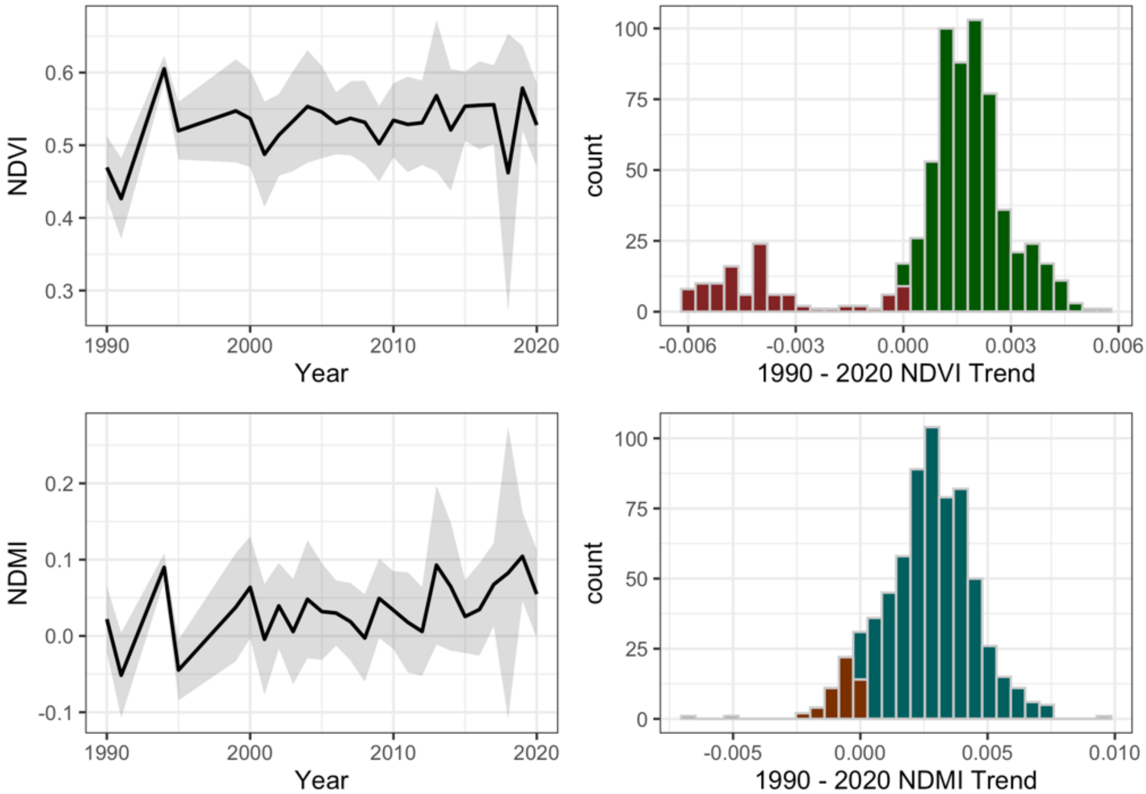

Figure 11. Average trajectories for NDVI and NDMI (left, not normalized) and distributions of trends (right) for NDVI and NDMI at ice-wedge degradation case study pixels (n = 679, across 15 sites) in northern Alaska. Trends were calculated as the slope from linear models fit at Landsat pixel for the vegetation index. Note that these trajectories were not normalized because ice-wedge degradation is an ongoing disturbance, rather than a single event.

3.3.3. Cryogenic landslides

Climate-induced thawing of permafrost-affected hillslopes can trigger a variety of abrupt and gradual disturbances involving the mass movement of soils, collectively termed “cryogenic landslides.” These landslides can result in losses of vegetation, followed by the development of successional vegetation on re-transported materials. Different forms of cryogenic landslides vary with respect to their spatial extent and temporal characteristics, and thus the pattern and rate of ecological succession after disturbance. These subtypes include 1) active-layer detachments, 2) frozen debris lobes, and 3) retrogressive thaw slumps.

Active-layer detachment slides are relatively small, local slope failures that develop after warm, wet summers, such that saturated active-layer soils slide abruptly over the permafrost table (Leibman 1995, Ermokhina and Myalo 2012, Verdonen *et al* 2020). Future climate warming and associated permafrost degradation, as well as increases in triggers such as extreme warm summer periods, increases in rainfall, and forest fires, could increase their frequency (Lewkowicz and Harris 2005). Frozen debris lobes are slow-moving, lobate permafrost features consisting of soil, rock, organic material, and ice that move down permafrost-affected slopes via shear along their bases (Darrow *et al* 2016, 2015, Simpson *et al* 2016). The distribution and dynamics of frozen debris lobes are comparatively poorly known.

Retrogressive thaw slumps are thermokarst slope disturbances that contribute large volumes of materials downslope to lakes, drainage networks, and coastal zones (Burn and Lewkowicz 1990, Lantuit and Pollard 2008). Initiation of retrogressive thaw slumps depends on local geomorphological conditions and meteorology. Fluvial erosion along riverbanks or coastal zones can initiate slope failures, promoted by extended warm and wet conditions (Burn and Lewkowicz 1990). Following an initial slope failure, exposure of ice-rich permafrost enables thaw slump development, which can persist for many years while the areal size of the thaw slumps can expand to tens of hectares. For example, the thaw slump shown in Figure 12b and 12c expanded from 0.63 ha immediately after the detachment failure in 2016 to 1.04 hectares three years later (Turner *et al* 2021).

The frequency and size of retrogressive thaw slumps can be highly variable within and among landscapes. The largest thaw slumps in North America have been observed in the Richardson Mountains and Peel Plateau regions, NWT, Canada (Lacelle *et al* 2015). This region, which includes the Mackenzie Delta, has experienced an increase in occurrences of thaw slumps in response to wet summer conditions (Lantz and Kokelj 2008, Kokelj *et al* 2015). Zwieback *et al* (2018) also found an increase in thaw slumps on the Tuktoyaktuk Peninsula, northwest of the Mackenzie Delta, Canada, and the Bykovsky Peninsula, Russia, associated with available energy and late-season rainfall. Many coastal areas have seen an increase in thaw slump activity, including Banks Island (Lewkowicz and Way 2019). Interactions with marine environments, including thermo-abrasion from waves and ice, can have a strong influence on thaw slump activity along coastlines (Günther *et al* 2013).

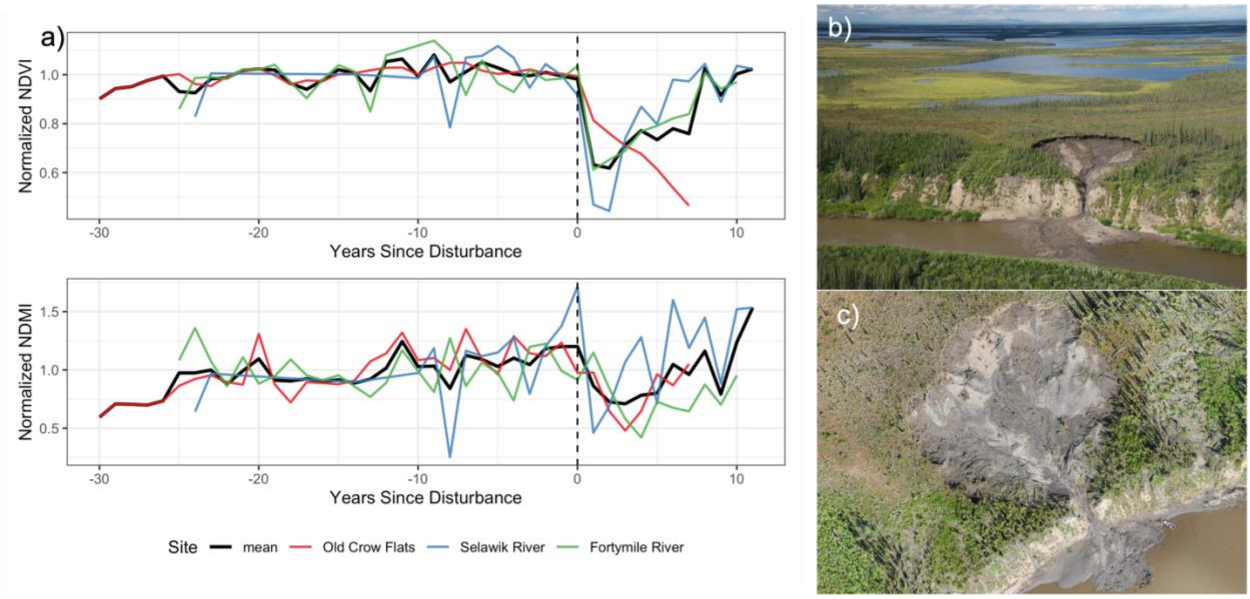

Figure 12. a) Case study trajectories for average (n = 3) and individual thaw slumps in Alaska (Selawik River) and the Yukon Territory (Old Crow Flats; Fortymile River); b) thaw slump in Old Crow Flats; c) aerial drone view of thaw slump in Old Crow Flats.

Cryogenic landslides impact terrestrial and aquatic ecosystems and atmospheric feedbacks. Within lake and river aquatic environments, biogeochemical cycling can be impacted by the liberated sediment and solutes, which are typically rich in nutrients and ions. However, the downstream impacts, on nutrient concentrations, for example, can be highly variable (Frey and McClelland 2009, Harms *et al* 2014, Lafrenière *et al* 2017, Mu *et al* 2017) and depend on local geomorphic conditions including relief, ice content, permafrost extent, and parent material (Tank *et al* 2020). These complex relations present uncertainties for associated impacts on local and downstream ecology. Vegetation can efficiently colonize stabilized areas of cryogenic landslides (Turner *et al* 2021). Habitat characteristics associated with landslide age and vegetation composition also have an influence on wildlife (Cray and

Pollard 2019). Atmospheric impacts include climate change feedbacks that stem from microbial decomposition of parent material and subsequent emission of greenhouse gasses (CO_2 and CH_4; Schuur *et al* 2015, Turetsky *et al* 2020, Miner *et al* 2022).

3.3.4. Lake drainage

Thermokarst lakes are formed when permafrost degradation results in subsidence of the land, which subsequently fills with water. These features are abundant across ice-rich permafrost terrain and are highly sensitive to climate conditions (Jones *et al* 2022). Though thermokarst lakes may remain stable for centuries, the shorelines are highly susceptible to erosion and expansion, the rate of which can be strongly influenced by dominant fetch and shoreline ground-ice content as well as climate (Roy-Léveillée and Burn 2010). When shoreline expansion progresses into low-lying areas or invades the boundaries of other thermokarst lakes, they can drain and experience near-complete water loss within days. Additional mechanisms that trigger drainage events can include drainage across an ice-wedge network, headward erosion along adjacent streams or coastal boundaries, and bank overflow when established outflow channels are blocked by snow and ice (Brewer *et al* 1993, Mackay 1981, 1988, Marsh and Neumann 2001, Hinkel *et al* 2007, Wolfe and Turner 2008, Jones and Arp 2015). Drainage can also occur incrementally through partial tapping by a stream and the development of open talik systems beneath the lake (Yoshikawa and Hinzman 2003).

Thermokarst lake drainage events represent drastic landscape transitions. Newly exposed lacustrine deposits serve as seedbeds for colonizing vegetation and can quickly develop continuous vegetation cover (e.g., *Eriophorum russeolum*, *Carex aquatilis*, and *Senecio congestus*) within the first few years following drainage depending on local conditions (Lantz 2017, Ovenden 1986, Mackay and Burn 2002, Shur and Jorgenson 2007). For example, willow (*Salix spp.)* encroached within 30.8% of the former 12 km^2 lakebed of Zelma Lake in Old Crow Flats, Yukon (Turner *et al* 2022). After a lake drainage event, the aquatic environment of the remaining water body can become highly dynamic for several years following drainage as biogeochemical properties are strongly influenced by weather and pluvial runoff across the exposed lakebed (Tondu *et al* 2017). Lake water biogeochemical properties stabilize as shrubs encroach, which enhances snowpack depth and snowmelt input (Turner *et al* 2022). The increasing NDVI in our lake drainage case study trajectories (Fig. 13) suggests encroachment of shrub vegetation. NDMI likely doesn't change because encroaching vegetation at these point locations are inundated with water.

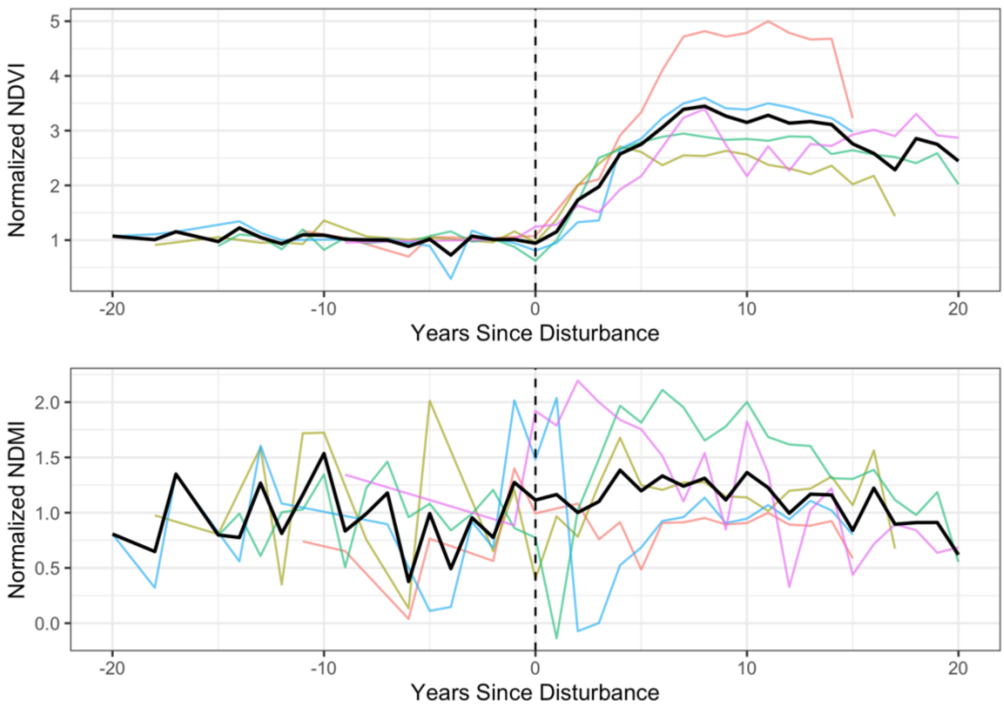

Figure 13. Average (black) and individual (colors) case study trajectories for lake drainage disturbance sites in Alaska and the Yukon Territory (n = 5) showing NDVI and NDMI normalized to the pre-disturbance average value.

Catchment hydrologic and vegetation characteristics typically do not return to pre-drainage conditions (Bandara *et al* 2020) and can thus exert long-term influence on carbon cycling. Drained lake basins can effectively sequester atmospheric carbon as peat accumulates (Fuchs *et al* 2019), though peat and carbon accumulation may eventually decrease (Bockheim *et al* 2004, Jones *et al* 2012, Fuchs *et al* 2019). Drained lake basins can remain dry for millennia (Shur and Jorgenson 2007, Hinkel *et al* 2003), and succession and ground-ice development may lead to variable species composition depending on local conditions.

Changes in the frequency of thermokarst lake drainage events have been highly variable among permafrost landscapes in Alaska (Jones *et al* 2011, Swanson 2019, Nitze *et al* 2020, Jones *et al* 2020a) and northwestern Canada (Lantz and Turner 2015). However, increasing temperatures and rainfall and associated increase in energy fluxes to permafrost will likely increase the vulnerability of thermokarst lakes to drainage (Turetsky *et al* 2020). In addition, lake drainage can be accompanied by the formation or expansion of other water bodies as observed in Siberia (Polishchuk *et al* 2015, Karlsson *et al* 2012, Nitze *et al* 2020), Alaska (Chen *et al* 2014), and the Tuktoyaktuk Peninsula (Olthof *et al* 2015, Marsh *et al* 2009). While the overall surface water area has remained stable in many of these regions, the spatial redistribution of water bodies suggests that these lake-rich landscapes are in a state of climate-driven transition (Rowland *et al* 2010, Pastick *et al* 2019). Ongoing research and monitoring will build our understanding of the short and long-term consequences for ecology, hydrology, and carbon cycling.

3.3.5. Limitations, data needs, and unknowns

Broadly, the study of permafrost-related disturbances would benefit from remote sensing studies which leverage higher-resolution sensors. Many of these disturbances at the individual scale can be quite small (e.g., frost circles, <3 m in diameter), and thus medium resolution satellites such as Landsat or Sentinel may miss small-scale changes in surface geology and vegetation driven by thermokarst processes. Additionally, more studies are needed to understand vegetation colonization and succession on newly available land created by permafrost-related disturbances.

Although frost circles are common across the entire Arctic climate gradient, the small size of individual features makes them difficult to detect, even in imagery with submeter spatial resolution. As a result, their distribution has not been mapped or constrained except at local scales. Such mapping of cryoturbated surfaces would be highly desirable, especially in the Low Arctic, where they are at risk of becoming less active (Aalto *et al* 2017). At present, areas that support dense frost circles can only be predicted based on coarse-scale maps of surficial geology and generalized soil texture. Whereas individual features may be challenging to identify, it may be possible to distinguish cryoturbated landscapes based on landscape-scale average spatial features.

There are numerous unknowns regarding the dynamics of ice wedge degradation and potential re-stabilization, and the extent to which they are occurring. Ice wedges are generally insulated by a mat of vegetation and accumulating snow in the winter, and it is still unclear what weather conditions induce ice wedge melting and what might drive heterogeneity in degradation among ice wedges. It is also unclear what factors drive vegetation succession following ice wedge degradation and the development of surface water ponds and troughs. One factor could be the availability of nutrients such as nitrogen and phosphorus (Beermann *et al* 2015, Herndon *et al* 2020), however only a few studies have attempted to address changes in nutrient concentrations following ice wedge degradation (Norby *et al* 2019, Kent *et al* in prep). Finally, the rates of accumulation of organic matter in degraded ice wedges and their potential for stabilization are still poorly understood. Field studies of ice wedge dynamics utilize space-for-time substitution, examining ice wedges at different stages of degradation (Jorgenson *et al* in press) as opposed to assessing the dynamics of individual ice wedges over time.

There has been substantial progress on our capacity to gauge the extent of ice wedge degradation utilizing high-resolution remote sensing and machine learning techniques

(Witharana *et al* 2021, 2020). Whereas these studies and associated applications can map ice wedge polygon networks across extensive areas of land, and even potentially estimate the fraction of land that contains ice wedges versus polygon centers, there is still work to be done to distinguish among the different stages of degradation.

There have been many studies that have documented the detection of cryogenic landslides (e.g, Balser *et al* 2014, Swanson and Nolan 2018, Barnhart and Crosby 2013), however, detection of the frequency of relatively small landslides may be difficult using medium resolution imagery (e.g., Landsat). Thus, large-scale mapping of these disturbances is difficult because the size of individual thaw slumps can be characteristically different depending on the region, and because high-resolution imagery at large scales is both cost prohibitive and difficult to work with.

Lake drainage events and associated impacts are complex and require additional research, especially where drainage frequency is increasing. Our ability to identify where and when thermokarst lake drainage will occur in the future must be refined. Existing data archives (e.g., Landsat 5 - 8, Sentinel-2) provide resources needed for identifying locations of past drainage and associated changes in land cover of larger lakes. While many studies have successfully utilized products from these sensors, the availability of scenes can be limited for any given year according to the timing of cloud-free conditions and the spatial resolution may not be adequate for detection of small-scale surface area change (e.g., < 30 m resolution) or for smaller water bodies. Broader coverage of high-resolution (optical, radar and elevation) products will improve these analyses and enhance detection of landscape responses to drainage and geomorphological characteristics (e.g., the proximity of lakes to low-lying areas) that make lakes vulnerable to drainage.

3.4. Anthropogenic disturbances

The North American ABZ has experienced extensive industrial activity and development in the last half-century (Schneider 2002, Pasher *et al* 2013). These disturbances include flooding for hydroelectricity, timber harvest and other natural resource development (e.g., mining, oil, and natural gas), including associated infrastructure such as pipelines, roads, and seismic lines for resource exploration. Additional highly localized disturbances in this region include landfills and dumps for disposal of domestic and industrial waste. These disturbances do not always fully remove or eliminate vegetation and soil, but often result in highly fragmented landscapes, leading to significant changes in ecosystem composition, structure, and function (Pasher *et al* 2013). As climate continues to change, northward expansion of agricultural areas is expected in southern regions of the ABZ, resulting in lasting removals of natural vegetative cover (King *et al* 2018). Although the cumulative area disturbed by the combined activities is vast, the impact of past and present natural resource development on ABZ ecosystem function (e.g. carbon cycling; Strack *et al* 2019, Schmidt *et al* 2022) and services (Pickell *et al* 2014) has often been overshadowed by fire and insect outbreak due in part to data limitations.

3.4.1. Logging

Forest harvest activities are major disturbances in Canadian forests (Gauthier *et al* 2015), with 35 to 40% of the Canadian boreal forest under industrial harvest and management (Burton *et al* 2003, Venier *et al* 2014). Industrial-scale forest management and economic activity have been an important component of the southern and eastern Canadian boreal forest since the 1800's (Venier *et al* 2014). For example, the Canadian forest products industry harvested over 710,000 hectares (~143 million m^3) of forest in 2020 (National Forestry Database 2020). In these higher productivity and more easily accessible southern and eastern regions, coniferous evergreen species (e.g., spruce, fir and pine) and aspen dominate the landscape and are utilized for lumber, pulp, and paper (Burton *et al* 2003, Venier *et al* 2014). In comparison, timber harvest is less extensive in the Alaskan boreal forest (Potapov *et al* 2008), where managed forests are generally concentrated in areas with high-value sawtimber species, adequate road access, and proximity to milling facilities – mostly in southeastern Alaska (Morimoto and Juday 2018) and episodically within the interior.

Clear-cutting is the most common silvicultural method used in the boreal forest (Haggstrom and Kelleyhouse 1996, Burton *et al* 2003, Cyr *et al* 2009). It was initially justified as an adequate replication of stand-replacing natural wildfire (Bergeron *et al* 2002); however, post-treatment belowground conditions (e.g., soil depth, nutrient content) can substantially differ from those following wildfire (Simard *et al* 2001), ultimately impacting post-disturbance successional trajectories in unique ways (Nguyen-Xuan *et al* 2000). Additionally, the coarse woody debris left after wildfire generates habitat for songbirds and other species, but is largely absent from post-harvest landscapes (Morissette *et al* 2002). Finally, post-treatment planting can increase regrowth compared to post-fire regrowth (Dieleman *et al* 2020). This is evident in the NDVI and NDMI time series for our logging case studies (Fig. 14), which in general show a faster initial recovery than those for fire (Fig. 4), and aligns with prior research (White *et al* 2017). However, it should be noted that other work has found the opposite result wherein post-fire forests recover slightly more quickly than harvested areas (Bartels *et al* 2016).

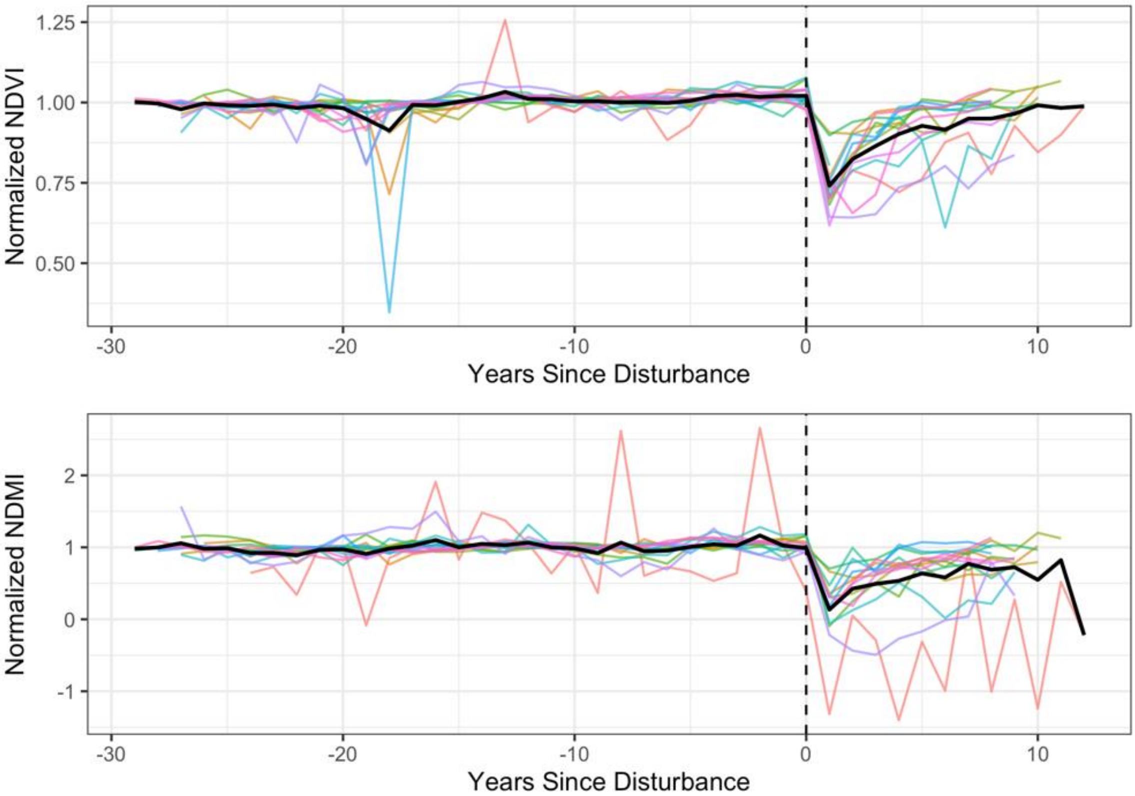

Figure 14. Average (black) and individual (colors) case study trajectories for logging disturbance sites in Saskatchewan (n = 14) showing NDVI and NDMI normalized to the pre-disturbance average value.

Traditional clear-cutting results in even-aged forest stands, as all trees are either harvested or disturbed due to harvesting activity, with only a small fraction left to stand as a seed source. Consequently, intensively managed landscapes often yield an even distribution of tree ages across the managed area, with no or few stands older than the harvest rotation time (Bergeron *et al* 2002). When the rotation time is shorter than the fire frequency, the resulting stands will be less diverse in terms of stand structure and species composition than stands that grow for longer periods and allow tree replacement or fires to kill a population of trees. Long fire intervals (e.g., 200+ years) allow for shifts in canopy dominance and forest age structure as a result of forest successional processes (Bergeron *et al* 2002). Thus, biodiversity concerns for highly managed areas have arisen, particularly in southern and eastern Canada (Venier *et al* 2014, Boucher *et al* 2009).

While clear-cut or group selection harvests predominate areas with high value stands or in areas where managers are attempting to mimic fire, approaches such as partial harvest and individual tree selection in mixed or deciduous stands often allow for individuals with a range of ages to coexist and the promotion of certain forest types (Gauthier *et al* 2009). Such harvesting practices can help increase diverse forest structural attributes, particularly in stands that are even-aged following prior harvest practices (Bose *et al* 2015). Comparatively, selective harvest is less impactful on total stand biomass than even-aged selection or fire, and thus has a more nuanced signal from remotely sensed data. Notably, many selective harvest practices, particularly those which promote specific species or are considered variable retention that retain structural elements of the stand, have been examined for impacts on avian (Schieck *et al* 2000), vertebrate (Vanderwel *et al* 2009), understory plant (Macdonald and Fenniak 2007), and beetle (Wu *et al* 2020) communities. While group selection and clear-cutting are most common throughout the boreal forests of the North American ABZ, harvest for the purpose of maintaining biodiversity or transitioning forest types for fire management (Astrup *et al* 2018) also occurs throughout the region. These different harvesting techniques and the degree to which outcomes can vary from technique to technique are an important component of the impact of forest management on boreal vegetation and soils, and warrants further study, especially in the context of ongoing shifts in climate and fire regimes that impact regeneration patterns.

3.4.2. Oil and gas well production

Oil and gas well production in the North American ABZ can be traced back a century to the still-active Norman Wells drilled in the 1920s in the Northwest Territories (Bone and Mahnic 1984). In British Columbia, the first commercial gas well was drilled along the Peace

River in 1947 and the first discovery of oil in Alaska occurred in 1957. The density of wells in the ABZ is typically less than 1 per km^2, however some locations can be as dense as 3 per km^2 or higher (Warrack *et al* 2021).

Over its long history, oil and gas exploration and the associated production technology, practices, and regulations have evolved (King and King 2013, Kang *et al* 2016). Depending on the type of well (e.g., conventional oil, unconventional gas), intent of the well (e.g., production, exploration, injection), geology (including depth and formation properties), and other factors, the resulting disturbance to the surrounding vegetation can be highly variable in terms of size, shape, and form, with the area of influence ranging from tens to hundreds of square meters. The disturbance also varies temporally throughout the life cycle of the well from site preparation to plug and abandonment (Burnham *et al* 2012, Allen *et al* 2013).

For both exploratory and development (or production) wells, well site preparation includes constructing a well pad and access roads. The lengths of new access roads for well sites in the Wayne National Forest in Ohio are 8 - 30 km (USFS 2004), but the lengths of new access roads needed in the ABZ may be much longer (Wilkinson *et al* 2021, Pasher *et al* 2013). The well pad involves clearing land so that the drill rigs can be brought in. Wells meant for producing oil and gas are first cased with steel piping and cemented, and then the inside of the innermost casing is connected with the host rock containing oil and/or gas. These activities remove vegetation, degrade soils, result in loss of seed and bud stores (Pickell *et al* 2015), and lead to overall biodiversity and habitat loss (McDaniel and Borton 2002, Butt *et al* 2013, Northrup and Wittemyer 2013). Due to the impacts of well production on soil nutrients, hydrology, and seed sources, regeneration on well sites is slower than that following fire or forest harvest (Osko and MacFarlane 2001). Forest succession and regrowth and overall landscape recovery can thus take decades following oil and gas activity (Powers *et al* 2015, Chowdhury *et al* 2017). NDVI and NDMI responses to oil and gas wells (Fig. 15) show a clear decline in both vegetation moisture and greenness, with recovery lasting longer than 10 years.

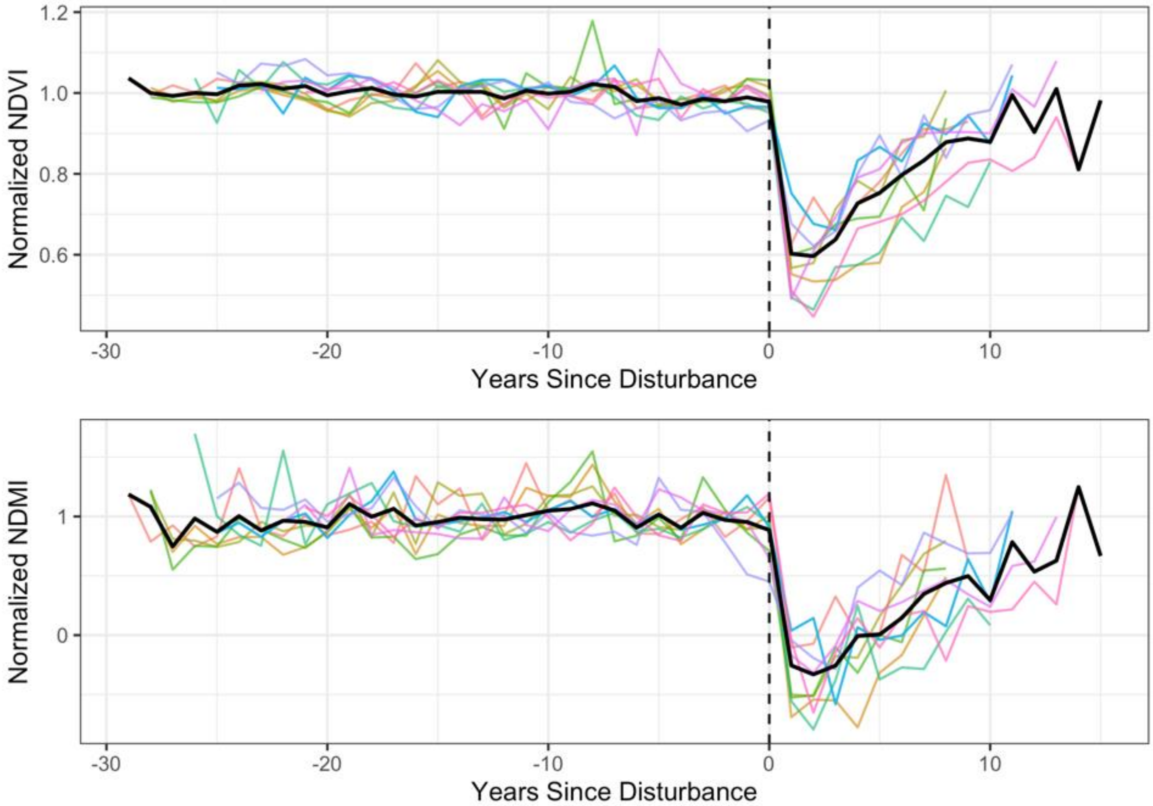

Figure 15. Average (black) and individual (colors) case study trajectories for oil and gas well sites (n = 10) showing NDVI and NDMI normalized to the pre-disturbance average value.

The production life of a well is highly variable, with some wells remaining in production for decades and others being abandoned after only a few years. Nevertheless, all wells are eventually abandoned and, according to modern regulations, must be plugged and the well site restored (Kang *et al* 2019). In Alberta and British Columbia, site restoration involves removing surface infrastructure and re-vegetating the land to pre-development conditions (Kang *et al* 2021). However, some wells have not been plugged and abandoned according to these regulations and have not had the surface restored.

3.4.3. Seismic lines

The largest anthropogenic disturbance across much of boreal and Arctic North America are seismic lines (Strack *et al* 2019, Jorgensen *et al* 2010), which are long linear clearings cut across forests and wetlands for oil and gas exploration (Fig. 1d, 16a,b). Seismic exploration for underground sources of oil and natural gas involves drilling a series of holes 6-20 m deep along the lines and analyzing the reflection of sound waves generated from either explosives detonated at the site or truck-mounted surface vibrators (EMR 2006). Originally, these lines (previously known as legacy or 2D lines) were cleared using heavy machinery to cut through heavily forested areas (Dabros *et al* 2018), creating lines up to 10 m wide. Individual length varies but combined create a vast network; Strack *et al* (2019) estimated 345,000 km of seismic lines crossing peatlands in Alberta alone. This type of clearing results in the complete removal of the aboveground woody vegetation (Filicetti *et al* 2019) and significant soil and peat compaction, causing the water table to be much closer to the ground surface (Davidson *et al* 2020b). These changes in soil characteristics and hydrological conditions can alter understory vegetation composition, including shifts from feather moss-shrub dominated understories to complete cover by sedges (e.g., *Carex aquatilis*) in fen peatlands, or sphagnum moss (*Sphagnum* spp.) in bog peatlands (Deane *et al* 2020, Davidson *et al* 2021). In recent decades, there has been a move towards a method called ‘low-impact’ seismic lines, created using lighter-weight machinery and by hand and allowing for minimal disturbance to the ground-surface (Dabros *et al* 2018). These lines are narrower (1 - 5 m) than legacy lines but they are far more abundant on the landscape, creating a dense grid-like network of disturbances and can still create substantial changes to both tree cover (van Rensen *et al* 2015) and understory vegetation communities (Davidson *et al* 2021).

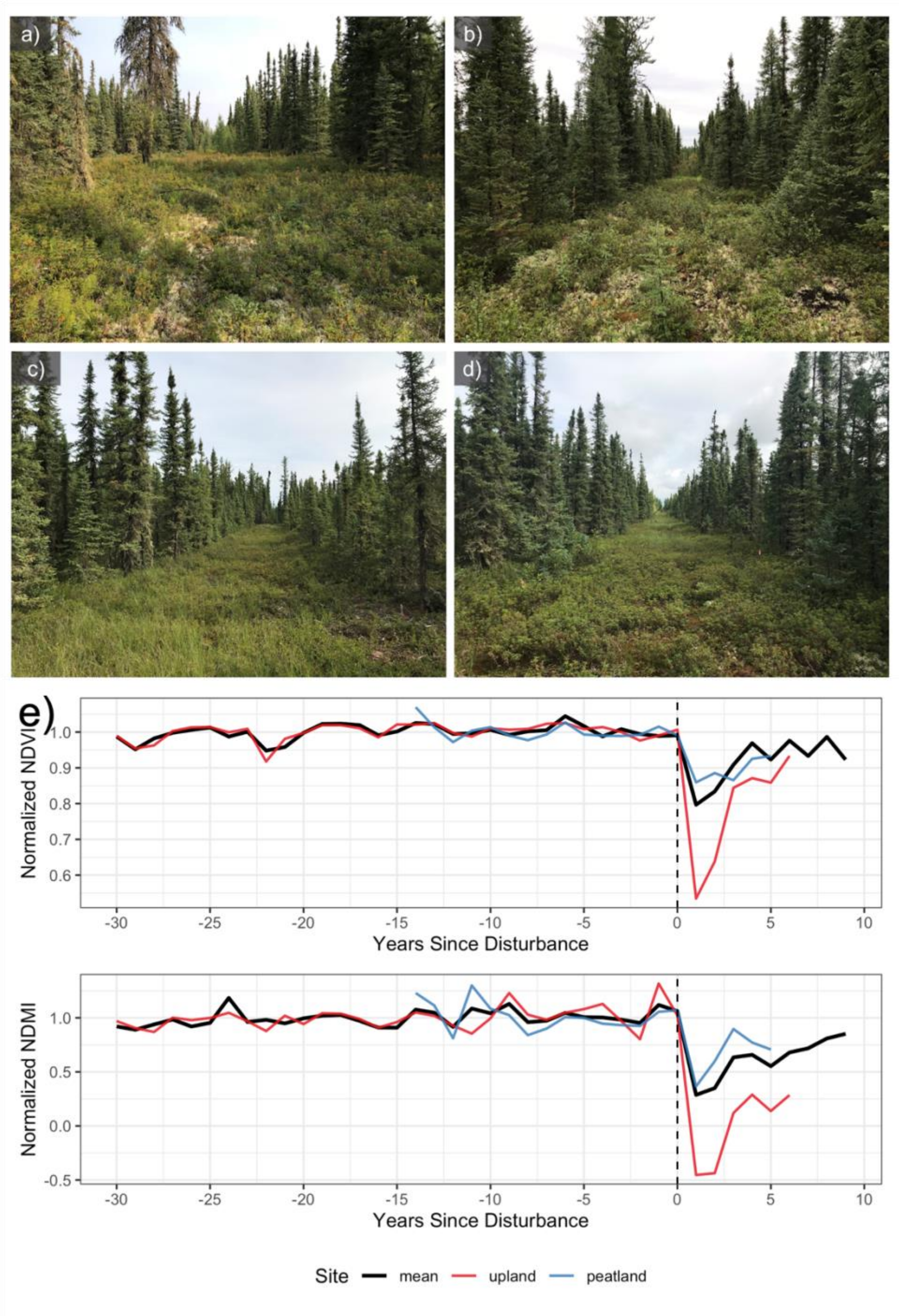

Figure 16. a, b) Seismic lines crossing upland boreal forest and c, d) peatland sites in northern Alberta, Canada. Note limited tree recovery on seismic lines crossing peatland ecosystems. All lines shown in these photos were cleared between 20-40 years ago; e) case study trajectories for average (n = 4) and two individual seismic line locations in Alberta, Canada showing NDVI and NDMI normalized to the pre-disturbance average value.

Although the creation of some seismic lines occurred almost 40-50 years ago, tree recovery and regeneration in many of these locations is slow and often fails. For example, Lee and Boutin (2006) estimated that after 35 years, approximately 65% of seismic lines crossing forests in Canada's boreal plains remained free of woody vegetation. Yet, our mechanistic understanding of how seismic testing influences vegetation recovery is limited. For example, in wetland locations, mechanical flattening of localized topography can result in a water table closer to the ground surface, leading to unfavorable conditions for black spruce (*Picea mariana)* seedlings to regenerate (Lee and Boutin 2006, Caners and Lieffers 2014). Furthermore, the post-disturbance understory vegetation communities, often dominated by hydrophilic species such as sedges and sphagnum mosses, may outcompete slow growing tree saplings (Davidson *et al* 2020b). In addition to the initial disturbance, continued use of these linear features for hunting, recreational sports, and further resource extraction activities can hinder tree recovery (van Rensen *et al* 2015). This poor recovery can be seen in our Landsat case studies of vegetation response to seismic lines (Fig. 16). There is a substantial drop in NDVI and NDMI at both upland and peatland seismic line sites following disturbance given trees are actively removed, and NDMI recovery is slow.

3.4.4. Limitations, data needs, and unknowns

Recent progress has been made to identify and map annual forest disturbance from logging across the North American ABZ based on the Landsat data archive spanning 1984 to 2014 (Zhang *et al* 2022). Between 1987 and 2012, 10.8% of the Alaska and western Canada experienced disturbance, with 1.4% attributed to logging. However, state and provincial forestry records are still an essential data source for understanding the scale and impact of logging and validating satellite detection of forest management, especially for lower-impact forestry practices that may be challenging for remote sensing approaches to detect. Such long-term data (e.g., polygons dating back to the 1960s, and GeoPDFs dating back to the 1800s in Saskatchewan, Canada) are crucial for studies of the impact of forest management on the North American boreal forest. However, many of these records are difficult to obtain.

Similarly, data and records of seismic lines are not readily available across all Canadian provinces.

There are limited studies on land disturbances caused by oil and gas well production and exploration with only a few recent studies that are based in the contiguous U.S. (Nallur *et al* 2020, Chomphosy *et al* 2021, Raynolds *et al* 2014). Though databases with information on wells (i.e., intent, type, age, etc.) are developed and maintained by numerous state, provincial, and territorial governments as well as the U.S. Bureau of Land Management for wells on federal lands, they can be incomplete (e.g., completely missing wells or incomplete information on well depth and age, etc.). Nevertheless, these databases have been compiled for Canada and the U.S. to understand oil and gas well distribution, methane emissions, and other environmental impacts (Kang *et al* 2021, Williams *et al* 2021). Commercial databases are also available (e.g., GeoScout), however, they are not likely to contain information on the size of well pads and land disturbances. There is research on using machine learning and high-resolution imagery to detect active oil and gas well pads, which may provide data on well pad sizes and shapes (Bartsch *et al* 2020). Overall, there is a need for improved oil and gas well databases and information on well pads to understand the full extent of impacts.

3.5. Weather-related disturbances

Though anthropogenic-driven climate change is likely to have longer-term “press” disturbance effects on ABZ vegetation, a handful of weather-related disturbances can affect vegetation markedly in the short-term, including rain-on-snow events, heat waves and extreme drought, and windthrow. Such disturbances can impact boreal and tundra vegetation, nutrient, and hydrology dynamics.

3.5.1. Rain-on-snow

Rain-on-snow events, or more broadly wet surface snow conditions (Pan *et al* 2018), are driven by a range of physical processes, though most often are caused by wintertime rain events that result in a wet snow surface (Singh *et al* 2000). Wet snow conditions can cause flooding and paludification in ABZ ecosystems, accelerate permafrost thawing, and decrease vegetation productivity (Rennert *et al* 2009, Bjerke *et al* 2014, Jeong and Sishama 2018). In mountainous regions rain-on-snow can destabilize the snowpack and trigger avalanches (Conway and Benedict 1994).

Most notably, re-freezing of melted snow creates ice barriers between the soil surface and the snowpack, making it difficult for ungulates such as caribou (*Rangifer tarandus*) and musk oxen (*Ovibos moschatus*) to forage for lichen during the winter (Putkonen *et al* 2009, Rennert *et al* 2009). These water and ice layers also facilitate the growth of toxic fungi, which can spoil lichens, further lowering wintertime food sources for ungulates, increasing foraging efforts and negatively impacting fat and protein reserves. In some cases, this can lead to movement of herds outside of their normal ranges, or even starvation and death, as occurred in 2003 on Banks Island, Canada (Putkonen *et al* 2009), when a severe rain-on-snow event resulted in the death of ~20,000 musk oxen, reducing the island's population by 25%.

Along with direct impacts of rain-on-snow on vegetation freezing and flooding damage (Bjerke *et al* 2015), such severe impacts on ABZ ungulates can have cascading impacts on vegetation, predators, and the human populations that depend on the herds (Serreze *et al* 2021, Sokolov *et al* 2016). A significant decline in ungulates in one region can potentially release that vegetation from grazing and trampling pressure, whereas a movement of ungulates into a new area driven by rain-on-snow may cause significant vegetation damage (Vors and Boyce 2009).

Occurrence of rain-on-snow events depends on several factors, including air temperature, precipitation type, and extent and thickness of the snowpack (McCabe *et al* 2007, Freudiger *et al* 2014). Increases in energy flux to the snow surface, either through increasing temperature or increases in latent heat from rainwater, cause snowmelt as well as subsequent disruption of the insulative effect of the snowpack on the soil through increased liquid water content and increased energy flux to the soil (Rennert *et al* 2009, Kim *et al* 2015, Pan *et al* 2018). While an individual rain-on-snow event is generally short-lived – on the order of days – the subsequent impacts on soil hydrologic and thermal conditions can last months. The frequency of rain-on-snow is predicted to increase in the future in the ABZ due to rising temperatures (Ye *et al* 2008, Jeong and Sishama 2018, Pan *et al* 2018), with potential cascading impacts on hydrology, thermal conditions, ecosystem function, and ecosystem services.

3.5.2. Heat waves and extreme drought

Heat waves and extreme drought can damage ABZ vegetation, lower productivity, and cause vegetation mortality (Hogg *et al* 2008, Allen *et al* 2010, Michaelian *et al* 2011). Heat waves can occur both during the growing season and in winter, with differing impacts on vegetation. Wintertime heat waves occur when temperatures rise above freezing for several days (Phoenix and Lee 2004, Bokhorst *et al* 2011). As a result, snow melts across large regions (Bokhorst *et al* 2008, 2009), initiating spring-like physiological responses in plants such as de-hardening and loss of frost tolerance, increases in photosynthesis, and bud swelling (Crawford 2008, Bokhorst *et al* 2010). Once temperatures return to freezing or below, plants are exposed to extreme cold due to reduction of snow's insulating capacity and buds can be damaged by frost (Bokhorst *et al* 2008, Girardin *et al* 2022). When the warming event is accompanied by little or no soil thaw, plant damage can be worsened by plant transpiration in frozen soil, leading to cavitation and desiccation of leaves, i.e., "frost drought" (Bokhorst *et al* 2008, Bjerke *et al* 2017, Comeau *et al* 2019). This plant damage can decrease productivity and lead to mortality. For example, an experimental manipulation study of a sub-Arctic heathland found a 50% reduction in GPP after multiple extreme winter

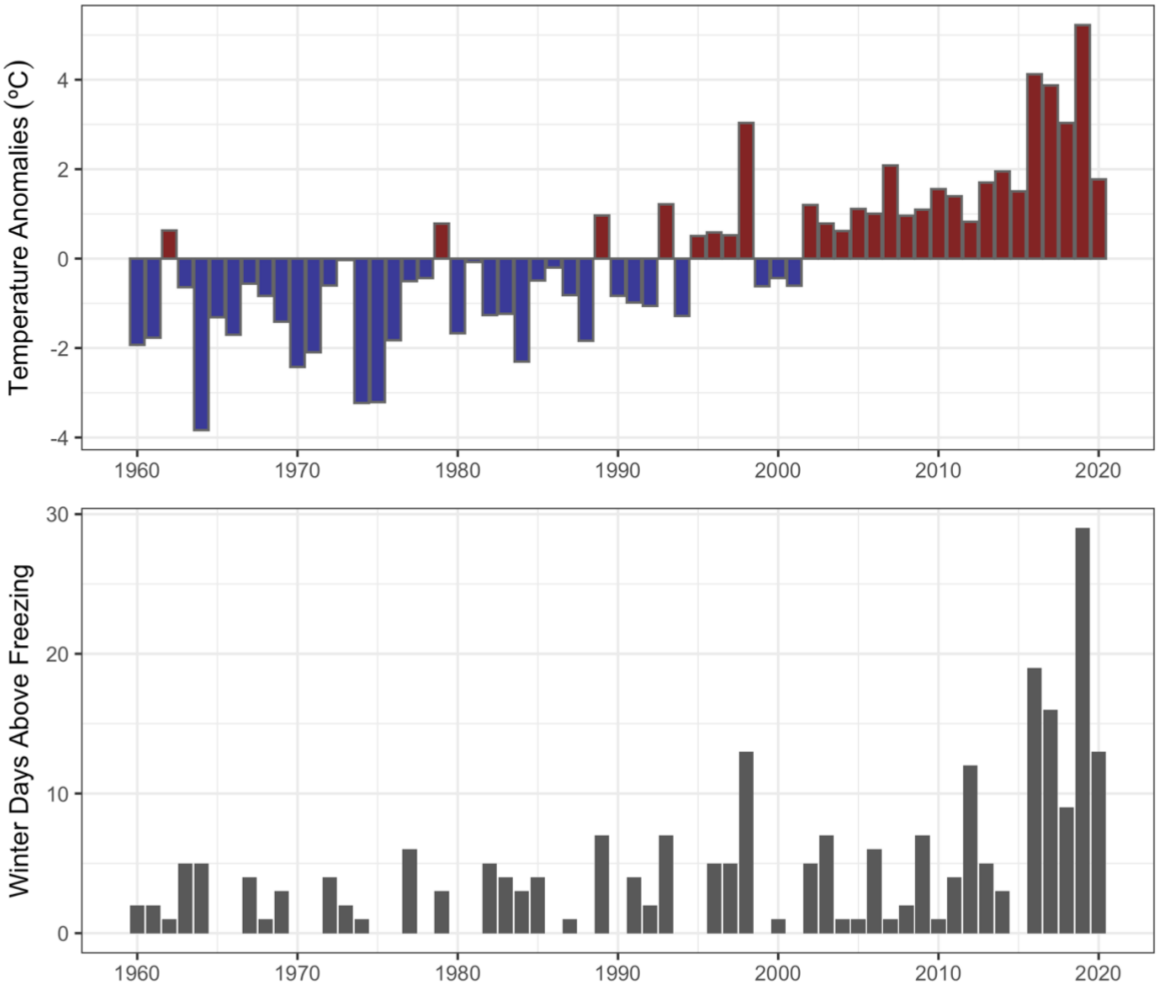

Figure 17. Climate data and temperature anomalies at Utqiagvik, AK (formerly Barrow, AK). Top: mean annual differences from 1960-2020 mean. Bottom: number of winter (Oct. – Mar) days above 0ºC each year from 1960-2020. Weather station data are from the NOAA Global Historical Climatology Network.

warming events (Bokhorst *et al* 2011). Such extreme warming events are predicted to increase in the future as temperatures rise (Meehl and Tebaldi 2004). In Utqiagvik, Alaska, the number of winter days with maximum temperatures above freezing has steadily increased since 1960, and several record-high days occurred in 2020 (as compared to the previous 20 years) (Fig. 17). Such temperature anomalies will continue to impact ABZ vegetation, potentially leading to plant damage and decreased productivity if wintertime extremes continue to increase (Richardson *et al* 2018).

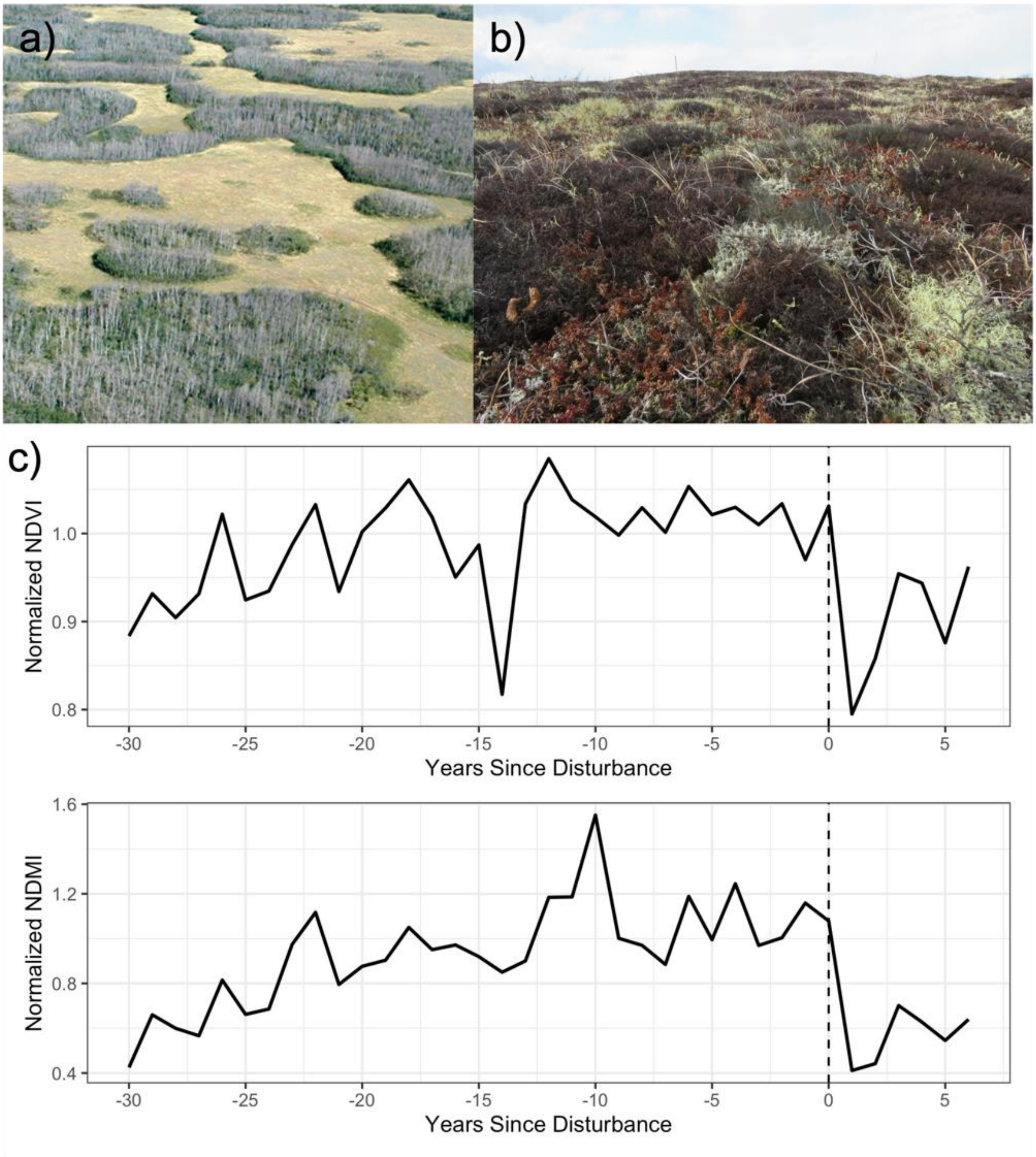

Figure 18. a) Massive mortality of quaking aspen in Saskatchewan, Canada, from a drought in 2001-2002, photo credit M. Michaelian 2004 (Michaelian *et al* 2011); b) browning of tundra vegetation; c) Landsat-derived NDVI and NDMI over vegetation in response to drought in the Northwest Territories in 2018.

During the growing season, heat waves and severe droughts (either from increased temperatures or decreased precipitation) can lead to water deficits that increase vegetation stress, lower productivity, and can cause widespread mortality under severe conditions (Fig. 18a; Hogg *et al* 2008, Allen *et al* 2010, Michaelian *et al* 2011, Peng *et al* 2011, Girardin *et al*

2021, Refsland and Cushman 2021). Such drought stress disrupts plant cell membrane function and can lead to xylem cavitation, with susceptibility to cavitation increasing with canopy height and varying by plant species due to differences in stomatal regulation (McDowell and Allen 2015, Allen *et al* 2010). Species-specific differences in drought response can alter stand structure and species composition if previously dominant species die off and are replaced by more drought-tolerant ones (Anderegg *et al* 2012). Drought can also cause regeneration failure and conversion of forests to woodland or grassland, particularly if compounded by other disturbances like fire (Baltzer *et al* 2021, Whitman *et al* 2019). Such drought and heat wave events can also trigger disease and insect outbreaks within already stressed vegetation (Raffa *et al* 2008, Boyd *et al* 2021, Ruess *et al* 2021). The impact of drought on vegetation greenness and moisture can be seen in a case study in the Northwest Territories for a drought that occurred in 2018 (Fig. 18c). Both NDVI and NDMI drop immediately following the drought, with slow recovery in vegetation moisture and more moderate recovery in NDVI.

Severe droughts and heat wave events are increasing within the North American ABZ, particularly in the southern boreal zone (Michaelian *et al* 2011, Berner and Goetz 2022, Perkins-Kirkpatrick and Lewis 2020). An extreme drought in 2001-2002 in southwestern Canada resulted in a severe aspen mortality event, with 45 Mt of biomass lost, resembling the carbon impacts from a severe wildfire (Michaelian *et al* 2011). Drought and heat wave events impact water quality, nutrient availability, and biogeochemistry (Tiwari *et al* 2018, Houle *et al* 2016) They also have the capacity to feed back to climate change through loss of carbon stocks and subsequent emissions from decomposition (Michaelian *et al* 2011, Ma *et al* 2012), as well as changes to energy and water cycling due to changes in surface roughness, transpiration rates, and latent heat fluxes (Bonan 2008).

3.5.3. Windthrow

Windthrow, or tree blowdown events from high wind, are important disturbance agents within the North American boreal zone that act primarily at the stand-scale (Ruel 2000, Bouchard *et al* 2009). While extreme wind events resulting in stand-replacement are rare in the boreal zone, partial windthrow where some individuals survive is more common, with return intervals ranging from 40 to 450 years in eastern Canada (De Grandpré *et al* 2018, Ruel 2000).

Damage to trees depends on individual tree and stand factors, including tree size, species, canopy position, and previous stem damage, as well as soil depth and moisture, stand density, fragmentation, and angle with respect to wind direction (Peterson 2004). Tree size and species are the most reliable predictor of windthrow survival – some tree species are more “wind firm” than others, and damage susceptibility increases with increasing tree size (Peterson 2004, Rich *et al* 2007). Because of the differential impact of partial windthrow on tree size and species, these events can cause shifts in the species composition and stand structure of impacted stands (Veblen *et al* 2001, Girard *et al* 2014). Windthrow can also act as a trigger for subsequent bark beetle outbreaks, as beetle populations are able to colonize and grow within downed stems (Wichmann and Ravn 2001).

3.5.4. Limitations, data needs, and unknowns

Some of the main challenges of studying extreme weather events like rain-on-snow, winter warming, and windthrow include the sparsity of weather stations in northern regions, the lack of routinely deployed weather equipment (Putkonen *et al* 2009), and the unpredictable occurrence of events such as severe blowdown (Bouchard *et al* 2009). Detection of rain-on-snow events with satellite measurements is possible using radar, microwave, and multispectral imagery (Serreze *et al* 2000, Pan *et al* 2018, Bartsch *et al* 2010). Accurate detection of windthrow depends on the spatial resolution of remotely sensed measurements compared to the scale of the blowdown (Schwarz *et al* 2003). An enhanced monitoring network of weather conditions and snowpack, such as those present in the SNOTEL network (Schaefer and Paetzold 2001) would help better characterize and identify the occurrence of these events.

With respect to extreme drought and heat waves, while the physiological mechanisms underlying plant drought response and vulnerability are well established and emerging remote sensing techniques offer promise (Rogers *et al* 2018), it is still difficult to predict which individuals will die from such drought stress (Trugman *et al* 2021). Critical needs include further understanding of plant physiological and site characteristics that influence drought exposure and susceptibility and better information about how biotic agents interact with drought to cause plant mortality (Trugman *et al* 2021).

3.6. Riverine processes

3.6.1. Background

Despite their relatively small footprint in ABZ landscapes, riparian zones are disproportionately important for ecological disturbance (Scrimgeour *et al* 1994), hydrological processes (Ploum *et al* 2021), biogeochemical cycling (Blackburn *et al* 2017), species diversity (Johansson *et al* 1996, Andersson *et al* 2000, Johnson and Almlöf 2016), and wildlife (Tape *et al* 2016, Cooke and Tauzer 2020). In recent decades, substantial hydrologic changes have been observed on ABZ rivers, including changes to seasonal flow-regimes (Peterson 2002, McClelland *et al* 2006, Smith *et al* 2007, Rawlins *et al* 2010, Holmes *et al* 2021), groundwater relations (Okkonen *et al* 2010), river-ice breakup (Prowse and Beltaos 2002, Beltaos *et al* 2006), biogeochemistry and water quality (Tiwari *et al* 2022), and beaver colonization (Tape *et al* 2018, 2022). In addition, there have been widespread changes observed in permafrost extent both on floodplains and within their catchments (St. Jacques and Sauchyn 2009, Jones and Rinehart 2010, Quinton *et al* 2011, Tananaev and Lotsari 2022). It has been hypothesized that these processes will lead to a reduction in the areal extent of active floodplains in ABZ landscapes due to increased river channelization, smaller peak flows, and reduced riparian disturbance intensity (Ström *et al* 2011, 2012, Nilsson *et al* 2013, Jansson *et al* 2019).

Streams and rivers in the ABZ are strongly influenced by geology and topographic relief as well as hydroclimate, ice cover, and the permafrost regime (Ashmore and Church 2001, Rokaya *et al* 2018), with high variability in river morphology (Nilsson *et al* 2015). Streamflow rates can range from slow-moving tundra streams to large flowing rivers that span Arctic-boreal ecotones (e.g., the Mackenzie and Yukon rivers) (Nilsson *et al* 2015). Riparian ecosystems are especially dynamic because they experience frequent erosion, flooding, and sedimentation (Wiens 2002). Channel migration and flooding can be seen as similar to fire disturbance, both creating short-term destruction to vegetation with the capacity for regeneration following the event (Rood *et al* 2007). Channel migration in particular can "reset" vegetation succession at any successional stage through floodplain erosion and simultaneous sedimentation and creation of new land for vegetation establishment (Walker and Chapin 1986, Viereck *et al* 1993, Van Cleve *et al* 1996, Helm and Collins 1997, Lininger *et al* 2017).

In addition to channel migration, ice-jam flooding is also an important disturbance in ABZ riparian zones. Ice-jams occur when ice floes in rivers are impeded by stationary ice covers, bridges, islands, or river width constrictions, leading to flooding (Rokaya *et al* 2018). Ice-jam flooding can occur during any river ice freeze-up or breakup period but are most common during the spring breakup period (Rokaya *et al* 2018, Beltaos and Prowse 2009). Ice-jam flooding causes significant economic and structural damage, and can result in loss of human life, made more prevalent by their unpredictable nature (Massie *et al* 2002, Mahabir *et al* 2008, Rokaya *et al* 2018). These floods also disrupt aquatic and riparian habitat through decreased fish habitat, and damage to and even removal of vegetation adjacent to the stream

(Lindenschmidt *et al* 2016, Lind *et al* 2014). Ice-jam flooding also exerts a strong influence on the water balance of lakes within river floodplains and deltas, and the floodwaters supply sediment, nutrients, and contaminants. These processes have been investigated in the Slave River and Peace-Athabasca Deltas where floodwaters replenish nearby basins and offset evaporative water loss (e.g., Brock *et al* 2009, Wolfe *et al* 2012) while also increasing concentrations of suspended sediment (and turbidity of the lake water), major nutrients, and contaminants such as polycyclic aromatic compounds and metals (Wiklund *et al* 2012, Hall *et al* 2012, Elmes *et al* 2016, MacDonald *et al* 2016, Kay *et al* 2020). Reductions in the frequency of flooding leave lakes across these landscapes at risk of drying (Wolfe *et al* 2012). Sustainable management of ice-jam flooding thus includes balancing both the detrimental and beneficial aspects of these events on socio-economic and ecological systems (Das *et al* 2018).

Beavers are important ecosystem engineers in the North American ABZ through their dam-building and hydrologic engineering of rivers, streams, sloughs, and lakes. Previously considered only a sub-Arctic species, recent observations show beaver colonization into low arctic tundra regions of Alaska and Canada in recent decades (Jones *et al* 2020b, Tape *et al* 2018, 2022) due to climate-change driven landscape change as well as population recovery from historical over-trapping (Tape *et al* 2018). Beaver dams trap water on the landscape, turning streams and sloughs into connected ponds, widening riparian zones and altering groundwater flow (Tape *et al* 2022, Westbrook *et al* 2006). Jones *et al* (2020b) found that beavers preferentially targeted thermokarst landforms in their dam-building activities within the Baldwin Peninsula, Alaska, accounting for 60% of the increase in surface water in the region between 2002 and 2019. Increases in surface and groundwater due to beaver dams transfers additional heat to the ground and thaws permafrost surrounding and beneath beaver ponds (Tape *et al* 2022). In permafrost-affected regions, beavers have the capacity to initiate and affect lake formation and drainage, ice-wedge degradation, cryogenic landslides, and other thermokarst events (Jones *et al* 2018, 2021). These physical changes to waterways and the surrounding permafrost effectively create warmer patches of mixed aquatic and terrestrial ecosystems that likely act as oases.

3.6.2. Limitations, data needs, and unknowns

Given the role of climate and extreme events on floodplains, spatiotemporal properties of disturbance, succession, and floodplain evolution are likely to be influenced by recent climatic warming at high latitudes, leading to important changes in the structure and function of riparian ecosystems in the ABZ. However, most ecosystem change studies to date have focused on upland and lowland ecosystems, whereas the observational record for riparian zones is comparatively sparse. There is thus substantial uncertainty concerning recent changes and future trajectories on floodplains across gradients of climate, stream order, catchment size, and floodplain morphology. For example, the pace of vegetation succession may increase in a warming climate due to longer, more productive growing seasons and changes in permafrost properties on or near riparian zones, particularly in forest-tundra ecotones (Wilmking and Juday 2005, Kharuk *et al* 2006, Beck *et al* 2011), while altered flow-regimes may influence the frequency and intensity of disturbance regimes. In Alaska, several studies have documented conspicuous, long-term increases in the extent and canopy height of tall shrublands in subarctic and Arctic riparian zones (Tape *et al* 2011, Brodie *et al* 2019, Liljedahl *et al* 2020). Understanding the interactions between biological and physical processes in the context of climate warning is important for assessing long-term impacts of continued warming on ABZ floodplains.

Beaver activity may be an important disturbance within permafrost regions, potentially causing widespread changes to the hydrologic and biotic environment, and initiating permafrost degradation (Tape *et al* 2022). Current research is exploring how these newly constructed oases affect carbon cycling, aquatic and terrestrial biodiversity, fish, and other ecosystem attributes. Further investigation is needed to understand the spatial extent and implications of beaver activity within the North American and circumpolar ABZ (Tape *et al* 2022).

3.7. Mammalian herbivore activity

3.7.1. Background

Mammalian herbivores like moose (*Alces alces*), caribou (*Rangifer tarandus*), and snowshoe and arctic hares (*Lepus americanus*, *L. arcticus*) impact ABZ ecosystems through coupled herbivore-vegetation feedbacks. For example, selective foraging, trampling, and inputs of excreta, urine, and decomposing carcasses can directly alter plant community composition or indirectly affect ecosystem properties through changes to soil characteristics and nutrient cycling (Leroux *et al* 2020, Olofsson *et al* 2004, Schmitz *et al* 2018, Väisänen *et al* 2014). These species are also a crucial subsistence resource for indigenous communities (Rexstad and Kielland 2006). Caribou in particular occur in high abundance across much of the North American ABZ, numbering in the millions, and are one of the Arctic's most ecologically, culturally, and economically important species (Gagnon *et al* 2020, Hummel and Ray 2008, Parlee *et al* 2018). These large herbivores also make some of the longest terrestrial animal migrations in the world, with some herds traveling over 1,000 km from boreal wintering grounds to Arctic tundra breeding grounds (Gurarie *et al* 2019, Joly *et al* 2019). During calving and migratory periods, caribou herds aggregate in dense groups and can alter landscapes as they pass through, impacting vegetation cover and structure, soils, and ecosystem carbon storage (Olofsson and Post 2018).

The distribution and intensity of caribou impacts are driven primarily by grazing and trampling associated with fluctuations in population sizes, which occur on a multi-decadal basis (Gunn 2003, Joly *et al* 2011, Vors and Boyce 2009). These fluctuations are influenced by snow conditions and forage availability (Gunn 2003, Joly *et al* 2011, Post and

Forchhammer 2002). A meta-analysis of caribou impacts on vegetation cover across the Eurasian and North American ABZ showed a clear negative effect on lichen (Bernes *et al* 2015). Because lichens are slow to recover from disturbance, this impact is both acute and long-lasting (Joly *et al* 2009, Macander *et al* 2020, Suominen and Olofsson 2000). Reductions in lichens in turn drive density-dependent feedbacks on caribou, causing population declines and influencing population cycles (Gunn 2003, Manseau *et al* 1996). Impacts of caribou trampling and grazing on vegetation can also include transitions to graminoid dominated communities (van der Wal 2006), and constraints on deciduous shrub expansion (Bråthen *et al* 2017, Christie *et al* 2015, Olofsson *et al* 2009) or treeline advance (Bryant *et al* 2014, Munier *et al* 2010). Caribou impacts are most pronounced in arctic environments where population densities are highest. In the boreal zone, low caribou density likely minimizes impacts.

In contrast, herbivores like hares and moose in the boreal forest can shift the age distribution of the foraged species towards younger age classes (Butler 2003, Kielland *et al* 2006). Selective feeding can also shift species composition. For example, moose herbivory can cause a shift from palatable deciduous species towards unpalatable evergreen species (Kielland *et al* 2006, Pastor *et al* 1988). Recent work suggests that moose alter their behavior to favor dense canopy areas during increased summer temperatures, suggesting shifts in areas vulnerable to browsing under warmer conditions (Jennewein *et al* 2020). Whereas moose generally avoid evergreen species like white spruce (*Picea glauca*), snowshoe hares browse heavily on white spruce seedlings, especially during periods of high hare abundance (Rexstad and Kielland 2006, Angell and Kielland 2009, Hollingsworth *et al* 2010, Sharam and Turkington 2009). Snowshoe hare populations in Alaska and Canada exhibit cyclic dynamics, driven by predator population size and herbivore-vegetation feedbacks (Krebs *et al* 2018). During peaks that occur about every ten years, snowshoe hare browsing can alter vegetation composition and plant chemical defenses (Fox and Bryant 1984), suppress the succession of white spruce (Olnes and Kielland 2016), and curb treeline advance (Olnes *et al* 2018).

3.7.2. Limitations, data needs, and unknowns

Most studies of herbivore impacts on vegetation use exclosures to assess what happens when herbivores are removed from a system. However, responses of vegetation to increasing vs. decreasing grazing pressure are not equal (Olofsson 2006). For example, studies that examine the impact of increasing caribou herd size (typically observational) often report stronger impacts than experiments that exclude caribou and examine the impact of decreasing herd size (typically manipulative) (Olofsson 2006). Geographic disparities in research can also influence conclusions. For example, studies of caribou impacts on vegetation primarily come from Fennoscandia (Soininen *et al* 2021). This raises issues of transferability of results because ecological conditions are different. Most caribou in Fennoscandia are managed in domesticated or semi-domesticated herds that often occur at higher densities than wild herds in North America (Bernes *et al* 2015).

Results from remote sensing and modeling studies which attempt to capture the relationship between caribou population density and vegetation productivity have produced mixed results, with some studies reporting significant negative relationships (Campeau *et al* 2019, Yu *et al* 2017, Rickbeil *et al* 2015) and others reporting weak or non-significant relationships (Fauchald *et al* 2017). Recent work by Davidson *et al* (2020a) provides and extensive collection of animal tracking datasets that can be used to analyze climate-driven variation in animal movement and foraging activity. As remote sensing technologies improve, increasing spectral and spatial resolution of satellite imagery might bolster the ability to quantify herbivore impacts across space and time.

4. Temporal and spatial scale of disturbances

Disturbances in the North American ABZ notably occur across a wide range of spatial and temporal scales (Table 1). The spatial grain of individual disturbance events ranges from on the order of meters for individual patterned-ground features such as frost circles (Frost *et al* 2013) to 1,000s of square kilometers for large boreal "megafires" (Stephens *et al* 2014). Temporally, ABZ disturbances occur over the course of hours or days, such as windthrow, or over years, such as with drought (Michaelian *et al* 2011). Their return frequency for the same location also varies from a general one-time event, such as with lake drainage (Shur and Jorgenson 2007), to an annual occurrence, such as with cryoturbation (Frost *et al* 2018b). Post-disturbance vegetation recovery times also vary, on the order of years (e.g. rain-on-snow; Bokhorst *et al* 2011), to decades (e.g. wildfire; Amiro *et al* 2010, Kurz *et al* 2013), or not at all (e.g., oil and gas wells; Kang *et al* 2021). Finally, the intensity of the impact on ABZ vegetation varies from productivity changes (e.g., cryoturbation, pathogens; Frost *et al* 2013, Holsten *et al* 2008) to complete vegetation loss (e.g. wildfire; Rogers *et al* 2015).

The temporal and spatial scale of disturbance occurrence and recovery as well as the overall intensity of impact can also vary within disturbance and landscape types. For example, high severity boreal wildfires tend to be stand-replacing large-scale events lasting weeks or months (Sedano and Randerson 2014, Rogers *et al* 2015, Veraverbeke *et al* 2017), in contrast to smoldering fires, which can burn year-round and survive the winter (Scholten *et al* 2021). Spatially, the resolution of individual disturbance events can be quite small but can cover large extents in their overall scale of impact. For example, insect infestations occur at the individual tree scale, but can then spread to whole stands and landscapes (Raffa *et al* 2008). Similarly, though individual seismic lines cover only a few meters in area, their combined extent is vast across the North American ABZ (Jorgensen *et al* 2010).

Table 1. Spatial, temporal, and intensity characteristics of ABZ disturbances.

Disturbance Group	**Disturbance**	**Spatial Grain**	**Return Interval**	**Occurrence Timeline**	**Recovery Timeline**	**Impact/Intensity**
Wildfire	Wildfire	100s of km^2	Decadal to centennial	Weeks to months	Decades to centuries	Some to complete vegetation loss
Insects and pathogens	Bark beetles	Meters to hectares	Decadal to centennial	Months to years	Decades	Some to complete vegetation loss
	Defoliators and leaf miners	Meters to hectares	Annual to decadal	Months to years	Years to decades	Vegetation loss; productivity decline
	Pathogens	Meters to hectares	Annual to decadal	Months to years	Years to decades	Some vegetation mortality; productivity decline
Permafrost	Cryoturbation	Meters	Annual	Months	Years	Stress
	Ice-wedge degradation	Meters	Annual	Years	Years	Partial mortality
	Cryogenic landslides	Meters to hectares	Decadal to centennial	Days to years	Decades	Vegetation loss
	Lake drainage	Meters to hectares	Generally one-time event	Days to years	Years to decades, if at all	Vegetation encroachment
Anthropogenic	Logging	Hectares	Decadal to centennial	Months	Decades to centuries	Vegetation loss
	Oil and gas wells	Meters	One-time event	Years	None	Vegetation loss
	Seismic lines and pipelines	Meters to hectares	One-time event	Weeks to months	Decades	Vegetation loss; vegetation change
Weather-related	Windthrow	Hectares	Decadal to centennial	Days	Decades	Some to complete vegetation loss
	Rain-on-snow	100s of km^2	Annual	Days	Years	Productivity decline; flooding; loss of grazing animals
	Extreme drought and heat waves	100s of km^2	Annual to decadal	Months to years	Years to decades	Vegetation loss; productivity decline
Riverine	Channel migration	Meters to hectares	Annual to decadal	Days to months	Years to decades	Some to complete vegetation loss
	Ice-jam flooding	Hectares	Centennial	Days	Decades	Vegetation loss
	Beaver engineering	Meters to hectares	Decadal	Months	Years to decades	Some to complete vegetation loss

Herbivore activity	Herbivore activity	Hectares to 100s of km^2	Annual to centennial	Months to years	Years to centuries	Vegetation stress; vegetation loss

We compiled these spatiotemporal characteristics across disturbance types (Table 1) and analyzed how they vary using a principal component analysis (PCA). The results from our PCA analysis (Fig. 19) indicate the broad spread in the spatiotemporal characteristics associated with ABZ disturbances. The loadings for frequency and intensity and size and occurrence/recovery timeline are opposite one another, indicating negative correlation. In general, high-intensity events occur at a lower frequency than lower severity disturbances which only impact productivity (but not necessarily mortality) (Table 1; Fig. 19). Some of the overarching groups are clustered together in the PCA (e.g., anthropogenic, pests and pathogens, weather), whereas the permafrost-related disturbances span the entire range of the first two principal components.

Understanding spatiotemporal differences is crucial when detecting and studying these disturbances via remote sensing, or when including them in process-based models. Advances in Earth observation sensor resolution have improved the capability to characterize and monitor disturbances and their interactions. However, in the context of detection and monitoring of multi-disturbance landscapes, an integrative approach is necessary to extend knowledge about disturbance (or multi-disturbance) recovery processes across high-latitude landscapes. Integration with remote sensing typically implies validation against pre-and post-disturbance *in situ* data across whole landscapes, and often involves cross-sensor harmonization to extend temporal or spatial ranges. Synthesis of disturbance-related studies toward understanding disturbance processes and their interactions across such a broad and heterogeneous domain requires bridging of temporal and spatial scales across scientific disciplines (i.e., ecology, geology, hydrology, etc.) (Cavender-Bares *et al* 2022). The disturbance spatial grain and extent are particularly important, and should match the spatial resolution of the sensor (Senf *et al* 2017b, Duncan *et al* 2020). Sensor pixel size is known to affect the measurement magnitude, location, and geospatial congruence of disturbance hotspots and the characterization of the effects of disturbances on ecosystems (Cavender-Bares *et al* 2022). While some of the mid-resolution sensors like Landsat have long records and are capable of tracking trajectories, they may be limited to tracking only larger-scale disturbances because their pixel size (e.g., 30 m) is large relative to the sub-pixel of disturbances such as cryoturbation (~1-5 m) or the early stages of insect outbreaks.

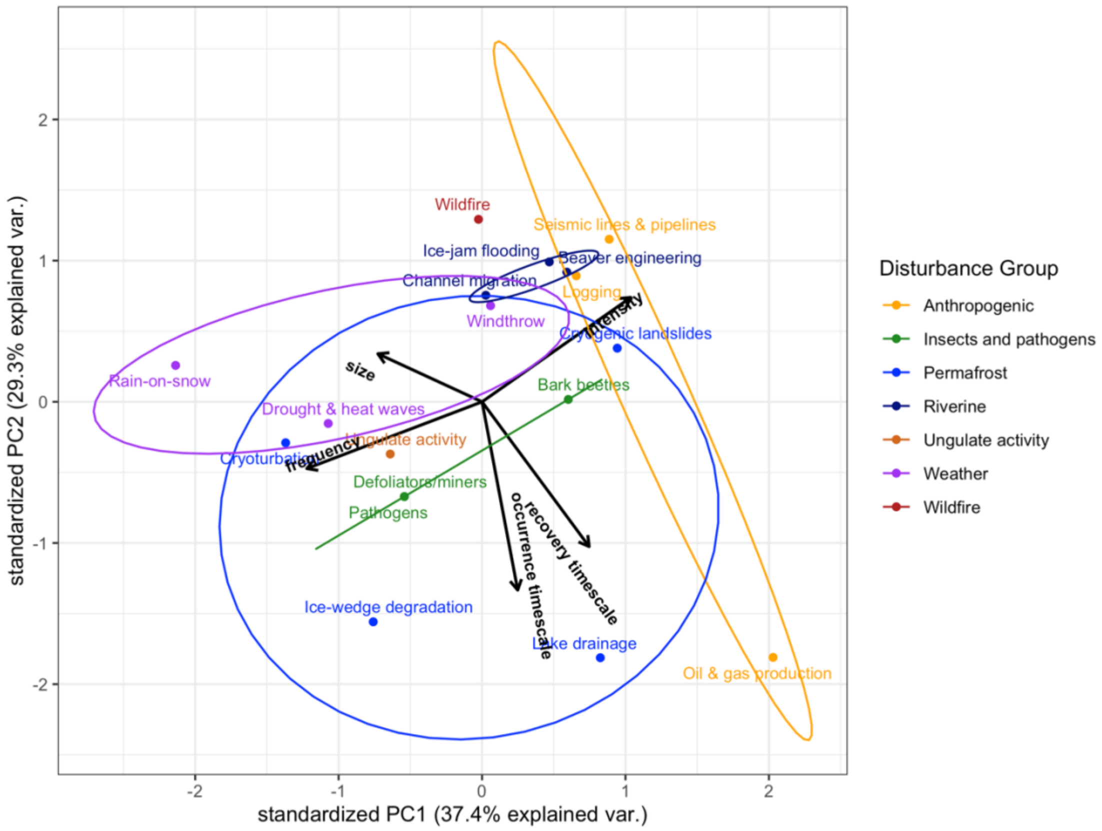

Figure 19. PCA derived from disturbance spatiotemporal characteristics (Table 1). Qualitative characteristics were modified to numeric scalar values (Table S2).

Scale is also crucial for the prediction of future disturbance effects, interactions, and feedbacks using process-based modeling. Models that do not consider individual plant species, such as many global climate models, will not fully capture species-specific effects of biotic disturbances, herbivory, and windthrow, or accurately capture successional dynamics following disturbances (Foster *et al* 2019, Shugart *et al* 2020). Ecosystem demographics represented in a modeling framework should interact with vegetation dynamically and be represented at scales that correspond to the frequency and extent of the disturbances that the model framework includes (Seidl *et al* 2011, Albrich *et al* 2020). For example, fine temporal scales (e.g., daily, Table 1) may be required to accurately model the disturbance interactions of a wildfire leading to a cryogenic landslide. It is also crucial to consider gridcell-to-gridcell spread of "contagious" disturbances like fire or insect infestation, as well as the temporal and spatial scales at which this spread occurs (Johnstone *et al* 2011). Representing the spatial and temporal complexities of multi-disturbance interactions in these systems accurately is an emerging area of high-resolution forest and tundra modeling. As remote sensing and modeling technologies improve, and more accurate and spatially continuous occurrence data are acquired, we will be better able to detect and predict ongoing ABZ disturbances, as well as their future trajectories.

5. Disturbance interactions

Disturbances within the ABZ can interact with one another, often with positive feedbacks that amplify the impact of subsequent events, such as wildfire and subsequent abrupt permafrost thaw (Gibson *et al* 2018). Other interactions may have a negative or dampening effect on subsequent disturbances, such as cryogenic landslides and subsequent reduction in wildfire potential (Fig. 20, 21). Broadly, disturbances may interact by altering the *resistance* of an ecosystem to subsequent disturbances, altering the probability of future disturbances, or by altering an ecosystem's *resilience*, or its ability to recover from a subsequent disturbance and its overall impact and severity (Buma 2015). As most of these disturbances are predicted to increase in frequency, severity, and/or extent with climate change (Veraverbeke *et al* 2017, Chen *et al* 2016, Pureswaran *et al* 2018, Turetsky *et al* 2020, Pan *et al* 2018, Berner and Goetz 2022), the opportunity for interactions among these disturbances will likewise increase, leading to potentially nonlinear and cascading impacts on ABZ ecosystems and vegetation (Buma 2015, Seidl *et al* 2017). Typically, studies of disturbances, in the ABZ or otherwise, only focus on a single disturbance type, and thus do not capture the true potential impact of a disturbance that includes its downstream effects on other disturbance regimes (Seidl and Turner 2022). Here, we discuss some of the interactions between ABZ disturbances and present our findings in Figures 20 and 21 but note that there are many complex interactions which are still the subject of further study.

Due to the ubiquitous nature of wildfire across the North American ABZ, fire interacts with most other disturbances within these regions (Fig. 20, 21). Drought and wildfire are often linked, with low moisture conditions increasing fuel flammability (i.e., decreasing resistance), and post-fire impacts on soil conditions often leading to moisture stress (i.e., decreasing resilience) (Whitman *et al* 2019, Baltzer *et al* 2021). In general, fire probability increases in the initial stages following bark beetle outbreaks as needles dry and thus become more flammable (Jenkins *et al* 2012, 2014). However, once the needles fall, the ground-to-canopy continuity is lost, thus lowering the probability of high severity crown fires. Low severity fires that damage trees but do not kill them can increase susceptibility to insect and pathogen attack and subsequent mortality (Hood and Bentz 2007), however stand-replacing wildfire removes host availability and thus decreases the risk for outbreak (Veblen *et al* 1994). Fire in permafrost areas can lead to thermokarst features, permafrost degradation, and changes to hydrology (Holloway *et al* 2020). Research indicates that tundra fires are becoming more frequent (French *et al* 2015, Hu *et al* 2015) and that post-fire deciduous shrub expansion may, in turn, further facilitate fire (Gaglioti *et al* 2021, Higuera *et al* 2008, Bret-Harte *et al* 2013, Lantz *et al* 2010a). However, herbivory and trampling of expanding deciduous shrubs has the potential to provide a negative feedback effect that lengthens fire return intervals in the Arctic (Christie *et al* 2015, Olofsson *et al* 2009, Bråthen *et al* 2017).

Beavers have also been shown to prevent fire spread and provide fire refugia (Fairfax and Whittle 2020).

Aside from fire, windthrow is often a precursor to bark beetle infestation through facilitation of beetle population growth within downed logs (Christiansen *et al* 1987, Malmstrom and Raffa 2000). Defoliators and bark beetles influence one another, where defoliators can weaken hosts and increase susceptibility to subsequent attacks by bark beetles (Cole *et al* 2022). Likewise, drought and biotic disturbances can enhance one another through decreased vegetation resilience (Malmstrom and Raffa 2000, Ruess *et al* 2021, Boyd *et al* 2021).

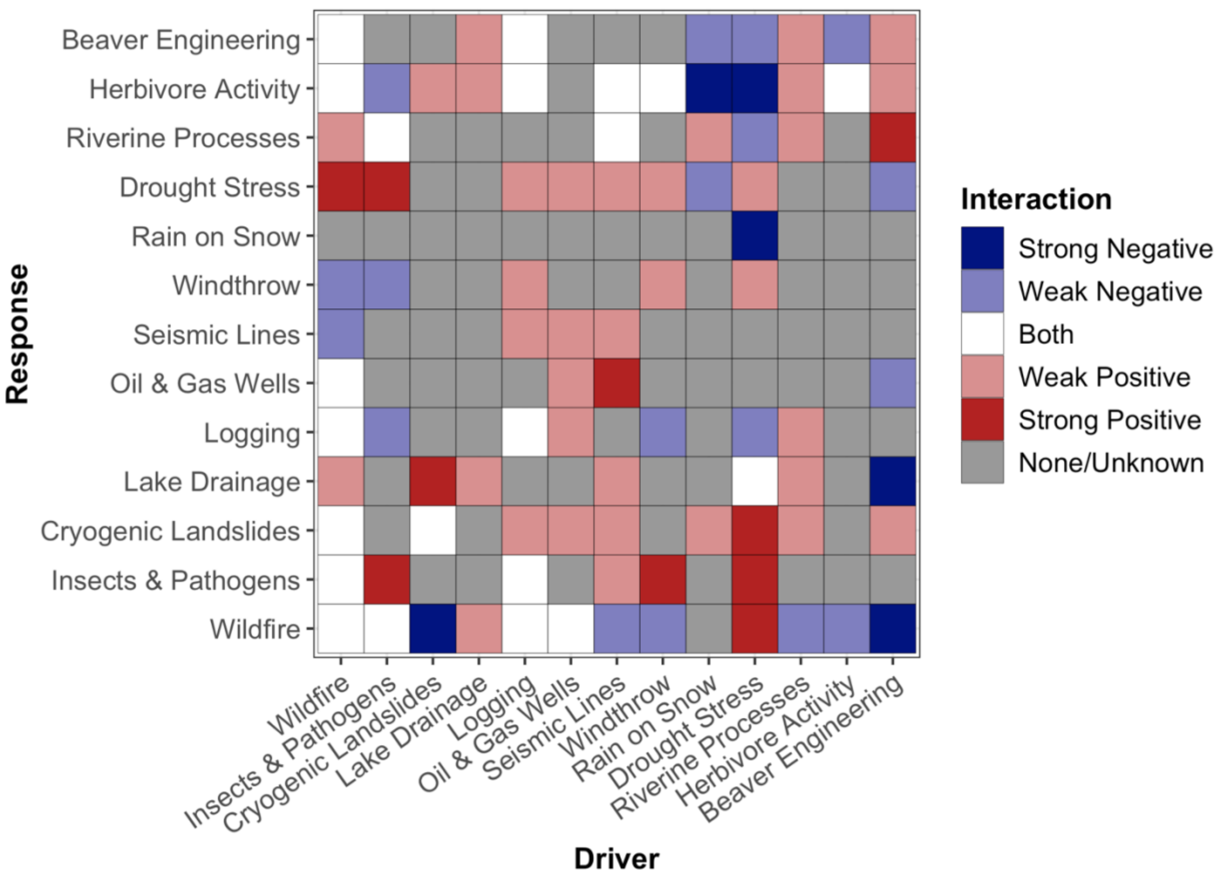

Figure 20. Interactions between disturbances in the North American boreal forest. Driver (x-axis) disturbances are the initiating disturbance, whereas response disturbances (y-axis) are the potential subsequent disturbances. Negative interactions correspond to a dampening effect of the driver on the response disturbance. Positive interactions correspond to an enhancing effect of the driver on the response disturbance.

Many disturbances are linked with cryoturbation, ice wedge degradation, and cryogenic landslides (Fig. 20, 21). For example, a physical disturbance to the landscape, such as a fire or seismic line placement, can reactivate cryoturbation features and local permafrost degradation by removing live vegetation and surface organic material (Frost *et al* 2013). Thaw slumps can also trigger catastrophic drainage of adjacent thermokarst lakes (Marsh *et al* 2009).

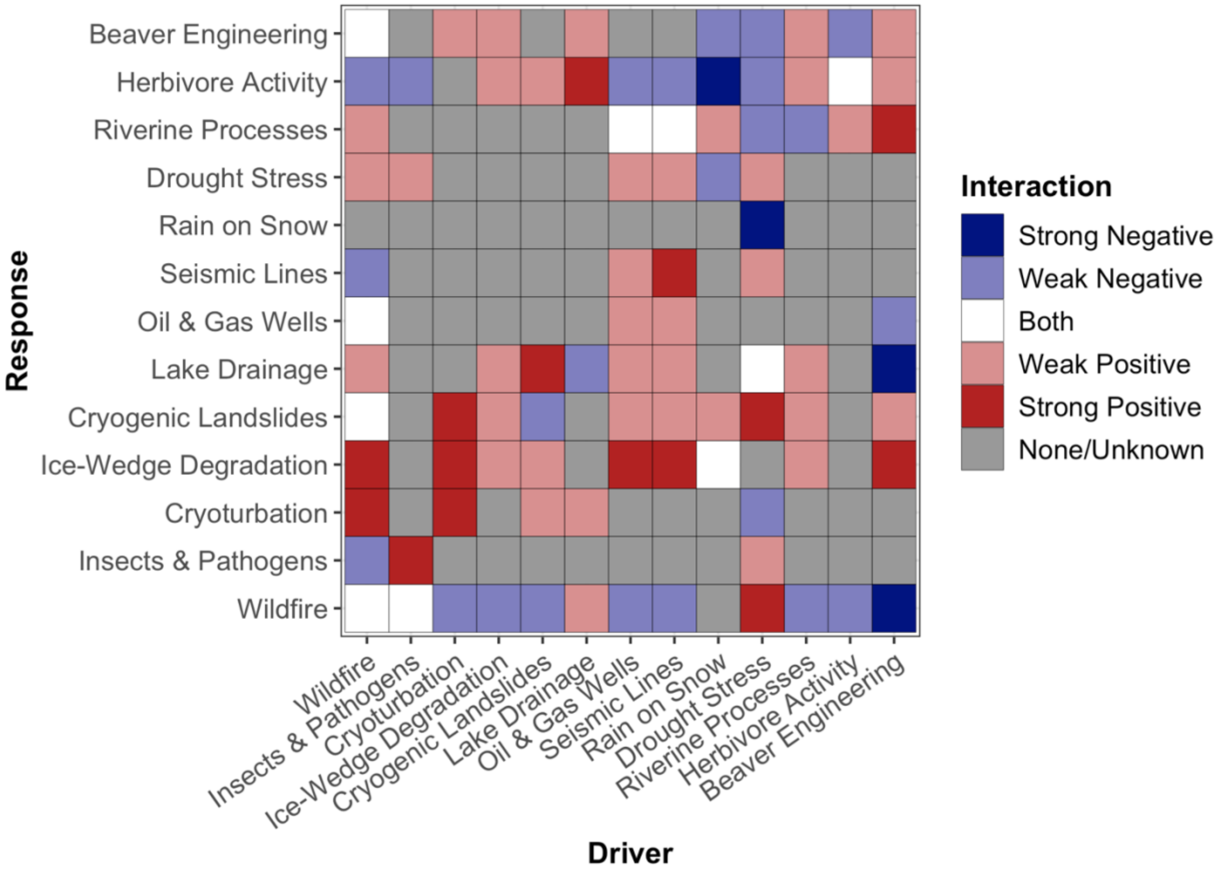

Figure 21. Interactions between disturbances in the North American Arctic tundra. Driver (x-axis) disturbances are the initiating disturbance, whereas response disturbances (y-axis) are the potential subsequent disturbances. Negative interactions correspond to a dampening effect of the driver on the response disturbance. Positive interactions correspond to an enhancing effect of the driver on the response disturbance.

Anthropogenic features such as roads, seismic lines, and logging affect the landscape and can result in additional disturbance; roads can lead to additional wildfires by opening access to human ignitions. Across Canada, the majority of human-caused ignitions are within

10 km of communities (Parisien *et al* 2020). These fires then have the potential to destroy human infrastructure. However, roads and infrastructure can also act as fire breaks and prevent fire spread (Cochrane *et al* 2012, Narayanaraj and Wimberly 2011). Some salvage logging can take place after a fire event, but fires can also destroy stands designated for harvest, or previously harvested stands. Insect outbreaks and pathogens have destroyed merchantable timber across Canada (Volney and Fleming 2000, Hennigar *et al* 2007), reducing the area available for harvest. The large network of seismic lines associated with oil and gas exploration has also negatively impacted habitat quality for boreal woodland caribou across Canada, with many populations in decline (Hebblewhite 2017, Nagy-Reis *et al* 2021). This type of habitat fragmentation has been shown to alter animal behavior and reduce mammalian movements globally (Tucker *et al* 2018, Finnegan *et al* 2018).

Many of the disturbances have no or unknown interactions (Fig. 20, 21), either because of lack of study (e.g., insect outbreaks and pathogens and subsequent thaw slumps) or because the disturbances are not generally co-located (e.g., cryoturbation and logging). These unknowns present both an opportunity and need for further study as well as the potential for previously geographically separate disturbances to interact as climate change continues to modify their extent and range. Disturbance interactions in particular should be a priority for further field, remote sensing, and modeling studies in the ABZ.

6. Conclusions

Present in all these disturbances is the amplifying effect of climate change, as this region is warming much faster than other areas of the globe (Price *et al* 2013, Smith *et al* 2019, Chylek *et al* 2022). The direction and magnitude of precipitation change is of growing concern, and this shift will feed back to changes in disturbance trajectories – a drier landscape will lead to larger and more severe wildfires, whereas abrupt permafrost thaw may increase in a wetter environment that dampens wildfire risk. Ultimately, disturbances are pivotal in creating local hotspots of change against the backdrop of long-term climate change. These disturbances create the potential for persistent shifts in vegetation composition (e.g., shift towards deciduous dominance post-fire) and biomass and extent (e.g., tall shrub and tree migration at treeline) (Foster *et al* 2022, Mack *et al* 2021, Maher *et al* 2021).

Disturbances also have the capacity to increase colonization and spread of non-native and invasive plant species (Kelly *et al* 2020, Sanderson *et al* 2012, Kent *et al* 2018). Previously, boreal and Arctic ecosystems were seen as too hostile and remote to facilitate invasion of non-native species (Sanderson *et al* 2012), however increasing temperatures and longer growing seasons are facilitating the northward migration of species in response to climate change (Chen *et al* 2011). Many studies have begun to document non-native and invasive plant species within the ABZ (Kent *et al* 2018, Wasowicz *et al* 2020, Leostrin and Pergl 2021), and show increasing establishment of these species following disturbances like fire (e.g., narrowleaf hawksbeard, *Crepis tectorum*, Carlson *et al* 2008) or harvest (e.g., bull thistle, *Cirisum vulgare,* Randall and Rejmánek 1993). Increasing anthropogenic presence and activities such as oil and gas exploration and production will also increase invasion of non-native plants, particularly in the Arctic (Wasowicz *et al* 2020). Through rapid growth, shading, and altered nutrient cycling (especially for N_2-fixing species) invasive plants can reduce growth of native plants, potentially leading to cascading impacts on biogeochemical cycling (Carlson *et al* 2008, Sanderson *et al* 2012). Though non-native and invasive species are gaining more attention in the ABZ, further studies are still needed to determine the potential pace of future colonization as well as how these species will interact with native flora in conjunction with climate change.

Table 2. Data needs and research opportunities for ABZ disturbances.

Disturbance Group	Disturbance Type	Data Needs and Research Opportunities
Wildfire	Wildfire	• More accurate, comprehensive, and finer scale burned area mapping

		• More combustion estimates • Post-fire vegetation trajectories and colonization of invasive/non-native species • Influence of forest and fire management • Future fire regime shifts
Insect outbreaks and pathogens	Insect outbreaks and pathogens	• Earlier outbreak detection • Accurate and spatially/temporally consistent datasets • Potential insect range shifts
Permafrost	Cryoturbation	• More accurate and finer-scale mapping
	Ice-wedge degradation	• Data distinguishing between different stages of degradation • Drivers of heterogeneity in degradation • Driver of vegetation succession following degradation
	Cryogenic landslides	• More accurate and finer-scale mapping
	Lake drainage	• More accurate and finer-scale mapping of drainage and associated impacts • Prediction of where and when lake drainage will occur in future
Anthropogenic	Logging	• More accurate and comprehensive records
	Seismic lines	• More accurate and comprehensive records
	Oil & gas well production	• More accurate and comprehensive records • Long-term impacts to vegetation and surrounding landscape
Weather-related	Rain-on-snow	• Enhanced monitoring networks • Cascading impacts on vegetation
	Windthrow	• Enhanced monitoring networks
	Drought and heat waves	• Better prediction of where, when, and which plants will succumb to drought mortality • Drivers of drought exposure and susceptibility
Riverine	Channel migration and ice-jam flooding	• More studies on riparian ecosystems in general • Vegetation succession in riparian ecosystems under climate change
	Beavers	• More beaver studies in general, especially in the Arctic
Mammalian herbivores	Ungulates	• More studies in North America on wild herds • Better data linkages between population size and satellite-derived vegetation response

Disturbances also interact with human society in fundamental and profound ways. Smoke from large fires in the ABZ can substantially reduce air quality (Trainor *et al* 2009, Johnson *et al* 2021), and fires themselves cause significant destruction of human property and resources (de Groot *et al* 2013, Thomas *et al* 2017). Many disturbances (e.g., insects, pathogens, windthrow, drought) reduce timber resources (Volney and Fleming 2000, Hennigar *et al* 2007, Anderegg *et al* 2012, Boucher *et al* 2018). Permafrost thaw and subsequent ground subsidence is hazardous for travel and can damage critical infrastructure (e.g., roads, airports, homes), with impacts across Alaska estimated to exceed $5 billion by 2099 (Melvin *et al* 2017a, Daanen *et al* 2012). Many indigenous communities depend on healthy caribou and other herbivore populations for subsistence, and these animals are central to many indigenous cultures (Gagnon *et al* 2020, Lamb *et al* 2022, Rexstad and Kielland 2006). Understanding how disturbance regimes and their interactions are changing is crucial for adapting human society to climate change in the rapidly warming far north.

These disturbances also have the capacity to feed back to further climate change through direct release of carbon dioxide and other greenhouse gases (Ueyama *et al* 2019), as well as aerosols and black carbon in the case of wildfire. Post-disturbance impacts on soil moisture, decomposition, and vegetation regrowth can feed back to climate through impacts on above- and belowground carbon stores, permafrost dynamics, and energy and water budgets (Randerson *et al* 2006, Ward *et al* 2012, Bonan 2008, Holloway *et al* 2020). Most of the ABZ disturbances discussed here are expected to intensify with a warmer climate (Veraverbeke *et al* 2017, Chen *et al* 2016, Pureswaran *et al* 2018, Turetsky *et al* 2020, Pan *et al* 2018, Berner and Goetz 2022), with a few exceptions: diminished cryoturbation is predicted as permafrost thaws and vegetation increases (Aalto *et al* 2017, 2021), and diminished fluvial disturbance is predicted along with diminished extent of active floodplain surfaces (Jansson *et al* 2019). Though most of these disturbances are natural and integral components of the ABZ system, anthropogenic climate change is pushing their extent, frequency, and severity outside of historical regimes. Continued study and data acquisition is crucial for projecting the future magnitude and direction of these disturbance trajectories and how they may interact (Table 2).

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