Post-fire recruitment failure as a driver of forest to non-forest ecosystem shifts in boreal regions

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Abstract

Climate change and land use are driving large changes in forest ecosystems around the globe. In the boreal biome it is likely that increases in temperature and the associated lengthening of the growing season will cause the forest to expand into the northern tundra and upwards in elevation, whilst potentially contracting at its southern limits. This increase in temperature is also driving an increase in the frequency and severity of boreal forest fires. A growing number of studies have observed the failure of forest species to re-establish after a stand replacing fire event which results in the shift to a non-forested ecosystem. In this chapter, this process is referred to as post-fire recruitment failure. We provide multiple lines of evidence for boreal forests, and more specifically for southern Siberia forests, that a possible regional tipping point is unfolding which could lead to the rapid replacement of large areas of forest ecosystems with low-stature non forest ecosystems. This change would come with significant consequences for the carbon balance, surface albedo, and the resulting altered energy balance.

1. Introduction

The boreal forest biome, also known in some regions as taiga, covers the high-latitude regions of Canada, Russia, China, Fennoscandia, as well as the United States, accounting for ~30% of all of the world's forested area (Gauthier et al., 2015) and ~20% of the world's terrestrial carbon sink (Bradshaw and Warkentin, 2015; Pan et al., 2011). These regions also have some of the largest areas of intact and unmanaged forest in the world (Potapov et al., 2017) and changes in this ecosystem can result in large-scale climate feedbacks (Chen and Loboda, 2018; Helbig et al., 2016; Liu et al., 2019). Boreal forests are dominated by a small number of slow growing tree species from four main genera, pine (*Pinus)*, aspen (*Populus*), larch (*Larix*) and spruce (*Picea.*) (de Groot et al., 2013; Rogers et al., 2015). Tree growth in boreal forests is strongly limited by air temperature with most growth occurring in the spring and summer months (Huang et al., 2010; Kauppi et al., 2014; Xu et al., 2013).

Over the last 100 years anthropogenic climate change has caused the boreal forest zone to warm at a much faster rate than most other terrestrial biomes (D'Orangeville et al., 2018) with this pattern expected to continue over the next century (IPCC, 2013). The warmer temperatures resulting from climate change have already resulted in increased vegetation productivity (Chen et al., 2016; Goetz et al., 2005; Kauppi et al., 2014; Keenan and Riley, 2018; Liu et al., 2015) and have driven the expansion of woody vegetation north into the tundra (Brodie et al., 2019; Forbes et al., 2010; Mekonnen et al., 2019; Myers-Smith et al., 2011; Suarez et al., 1999).

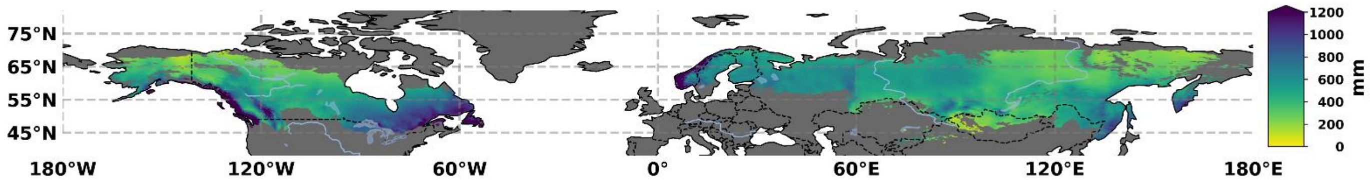

Figure 1 *Mean annual precipitation (mm) for the period 1958-2017 over the boreal forest. Non-boreal forest ecosystems are masked in grey. Data: TerraClimate (Abatzoglou et al., 2018a),the boreal forest mask (Potapov et al., 2008)*

While climate change is causing boreal forests to expand northward and upward, there is growing evidence that it is driving a contraction along the southern edge (Guay et al., 2014; Huang et al., 2010; Koven, 2013; Lapointe-Garant et al., 2010; Payette and Delwaide, 2003). It remains an open question as to whether the extent of the boreal forest zone will increase or decrease due to future climate change (Brown and Johnstone, 2012; Camill and Clark, 2000; Guay et al., 2014; Lapointe-Garant et al., 2010; Walker et al., 2019). Growth in southern boreal regions is often strongly limited by water availability, with parts of Siberia receiving less than 200 mm of rainfall per year (Figure 1). Droughts (Allen et al., 2010; Hogg et al., 2008; McDowell and Allen, 2015; Michaelian et al., 2011; Worrall et al., 2013), changes in fire regimes (Curtis et al., 2018) and direct human impacts (Svensson et al., 2019) have all been implicated in catastrophic forest loss (Achard et al., 2006) and the expansion of steppe vegetation especially in the southern boreal regions (Kukavskaya et al., 2016). Of particular concern is the change in fire regime, with some studies suggesting that increases in the frequency and severity of fires resulting in the permanent shift from forests to non-forested ecosystems which could in turn lead to a reversal of the boreal carbon sink (Bradshaw and Warkentin, 2015; Flanagan et al., 2016; Koven, 2013; Scheffer et al., 2012).

There is strong evidence that the boreal zone is experiencing large gains and losses in forested area (Hansen et al., 2013; Keenan et al., 2015). There is an urgent need to understand whether post-fire recruitment failure will cause ecosystem collapse in large parts of the boreal forest and its permanent replacement with grassland ecosystems. This chapter is broken down into three sections. The first discusses the important role that fire plays in boreal forests and the growing evidence for post-fire recruitment failure. The second section describes how climate change and forest management are driving changes in the boreal zone. The third discusses the difficult of quantifying the scale of post-fire recruitment failure and why this may be impairing our ability to predict future climate change. The final section outlines the management options.

2. Role of Fire in Boreal Forests

Forest disturbance can be defined as any period where the total amount of carbon in an ecosystem decreases beyond the normal interannual variability (Brazhnik et al., 2017). Wildfires are the dominant cause of disturbance in boreal forests (Bond-Lamberty et al., 2007; Curtis et al., 2018; de Groot et al., 2013; Goldammer and Furyaev, 2013; Hansen et al., 2013) with stand-replacing fire in particular shaping forest extent, composition, structure, and floristic diversity (Bonan and Shugart, 1989; Hart et al., 2019; Rowe and Scotter, 1973). Fire is a natural part of the boreal ecosystem (Johnstone et al., 2010) with palaeoecological reconstruction indicating that variations in fire frequency have been shaping species composition over the Holocene (El-Guellab et al., 2015; MacDonald et al., 1991; Novenko et al., 2016; Rolstad et al., 2017). Boreal forest fires are characterised by large areas of forest burnt in a single year followed by a gradual recovery (Curtis et al., 2018).

Globally, an average of between 10 and 20 million hectares of boreal forest burn annually, though there is considerable uncertainty around this estimate especially within Siberia (Brazhnik et al., 2017; Conard et al., 2002; de Groot et al., 2013; Flannigan et al., 2009, 2009; Kukavskaya et al., 2012; Rogers et al., 2020). There is also considerable interannual variability in the extent and severity of wildfires (Abatzoglou et al., 2018b; Beurs et al., 2018; de Groot et al., 2013; Kasischke and Turetsky, 2006), with the largest and most severe fire years associated with large-scale climate modes including the El Niño Southern Oscillation (ENSO), the Pacific Decadal Oscillation (PDO) and the Artic Oscillation (AO) (Balzter et al., 2007; Macias Fauria and Johnson, 2008; Monks et al., 2012; Ward et al., 2016). While decadal climate variability does lead to larger fire yeares, is should be noted that the source of ignition of almost 90% of fires is anthropogenic (Mollicone et al., 2006).

Fire intensity, size and mean fire return interval (FRI) vary considerably among different parts of the boreal zone (Archibald et al., 2018; Kharuk et al., 2011; Parisien et al., 2011; Sannikov and Goldammer, 1996). For example, in Siberia FRI decreases along a north-south gradient and more gradually from west to east (Abatzoglou et al., 2018b; Soja et al., 2006) whilst within North America, the FRI is shortest in central Canada and longer near coastlines and on the northern boundary of the boreal zone (Potter et al., 2020; Rogers et al., 2013). In the colder and wetter boreal regions, the FRI ranges from between 100 and >1000 years (Balshi et al., 2007; Kharuk et al., 2016; Kim et al., 2020; Mollicone et al., 2002; Schulze et al., 2005; Shuman et al., 2017). This follows a gradient through the warmer and drier parts of the southern boreal forest where the FRI is between 25 and 75 years (Chu et al., 2016; Rolstad et al., 2017) to the grassland and steppe vegetation that border the boreal forest, which have FRI of less than 17 years (Frelich et al., 2017). Similar patterns are apparent in the fraction of forest burnt each year (Abatzoglou et al., 2018b).

2.1 Post-fire recruitment dynamics

Given a stable climate and a time scale of centuries to millennia, boreal forests reach an equilibrium state where the amount of biomass lost to wildfire disturbances is balanced by the rate of recovery (Brazhnik et al., 2017) and where the species composition is adapted to the fire regime (Johnstone et al., 2010). In the colder and wetter parts of the boreal zone, the long FRI allows sufficient time for relay succession (Kurkowski et al., 2008; Ott et al., 2006). In these regions, the initial post-fire recruitment is dominated by broadleaf deciduous trees and then the ecosystem is overtaken by conifers over the course of centuries (Bergeron and Fenton, 2012; Kurkowski et al., 2008). In the warmer and drier parts of the boreal zone the seedlings that establish in the 1-5 year recruitment window after a stand-replacing fire event determine future canopy composition because the FRI's are too short to allow for relay succession (Johnstone et al., 2010; Moser et al., 2010).

Increases in the frequency and/or severity of the disturbance regime, or a decrease in an ecosystem's resilience to that disturbance, can trigger a tipping point that leads to complete state change in that ecological system (Brazhnik et al., 2017; Hart et al., 2019; Héon et al., 2014; Johnstone et al., 2010; Walker et al., 2019). There is growing evidence that the frequency, extent and severity of fires are increasing (Brazhnik et al., 2017; de Groot et al., 2013; Malevsky-Malevich et al., 2008; Rogers et al., 2020; Turetsky et al., 2011; Young et al., 2017) and that tree regeneration is decreasing (Stevens-Rumann et al., 2018) within the boreal zone. In some regions, this is resulting in a species composition shift from conifers to deciduous trees, as a result of intense fires destroying the aerial seed banks that conifers rely upon for rapid post-fire establishment as well as the shorter FRI which favours faster-maturing species (Hart et al., 2019; Héon et al., 2014; Johnstone et al., 2010; Johnstone and Chapin, 2006). These fire-induced species balance shifts have been observed in sites across the boreal zone (Héon et al., 2014; Johnstone et al., 2010; Johnstone and Chapin, 2006; Lara et al., 2016).

2.2 Post-fire recruitment failure

In the warmer and drier parts of the boreal zone, there is growing evidence of post-fire recruitment failure, with tree species failing to re-establish entirely (Stevens-Rumann et al., 2018). Whilst the window of opportunity for seedling establishment varies between regions, it is generally short (1–5 years) and the canopy composition is determined by the seedlings established during this period (Moser et al., 2010). Given the importance of this 5 year window (Johnstone et al., 2010), it is generally assumed that if tree species fail to establish in this time it will result in permanent forest loss and the expansion of non-forested ecosystems (Enright et al., 2015; Scheffer et al., 2012; Seidl et al., 2017). Whilst a documented lack of large-scale, very long term (>20 years), post-fire vegetation studies (Gitas et al., 2012) make the assumption difficult to confirm, it is consistent with some regional data.

A study of vegetation 30 years post-fire along the Canadian Alaskan border region found that the vast majority of recruitment occurred "*in the first 5 years after fire, and additional net establishment was not observed after 10 years*" (Johnstone et al., 2004). Similarly, an unpublished 2019 field survey of tree demographics at 16 long term (greater than 20 years) burn sites in Siberia found that >90% of Scots pine trees present at the sites established within 5 years of the burn (authors, unpublished). There is also evidence that analysis of existing canopy gaps in jack pine forests are driven by early poor regeneration density (Pacé et al., 2019), which supports the conclusion that recruitment failure in the first 5 years will result in permanent forest loss. Assuming that the 5 year establishment window is consistent in all regions, then post-fire recruitment failure and its associated permanent forest cover loss has been observed in Siberia (Barrett et al., 2020; Kukavskaya et al., 2016; Shvetsov et al., 2019), Europe (Moser et al., 2010), Canada (Payette and Delwaide, 2003; Splawinski et al., 2019; Whitman et al., 2018) and the United States (Frelich et al., 2017; Hansen et al., 2018; Stevens-Rumann et al., 2018). Whilst recruitment failure has been observed in multiple studies throughout the boreal zone, almost all of the existing literature (including the studies cited above) is focused upon understanding and quantifying species balance shifts between forest types rather than on a shift to non-forested ecosystem types. This means that empirical quantification of post-fire recruitment failure are *"scarce, and their underlying processes are not well understood"* (Boucher et al., 2019).

Despite the above, it is known that there are some factors that cause recruitment failure that have hard thresholds beyond which recruitment failure is almost certain (Hansen et al., 2018; Kukavskaya et al., 2016). These factors include post-fire water availability, the FRI, the disturbance history and the distance to seed source. The most common of these is FRI, with multiple studies from throughout the boreal zone finding that a short interval between stand-replacing fires (≤20 years) will almost guarantee recruitment failure (Brown and Johnstone, 2012; Kukavskaya et al., 2016; Whitman et al., 2018). However, in other regions the cause of post-fire recruitment is multifaceted and related to a *"resilience debt"* (Johnstone et al., 2016) which makes it difficult to ascribe attribution in the absence of extensive site histories. An excellent example of this issue comes from a study of managed forests in Canada (Perrault-Hébert et al., 2017). The latter authors found that a site had a ~50% chance of experiencing post-fire recruitment failure if it had had undergone logging at any point in the previous 50 years, which is almost double the rate of non-logged sites. The lack of data means that many of the recent attribution studies have used modelling to determine the causes of post-fire recruitment failure (Boucher et al., 2019; Splawinski et al., 2019; Stevens-Rumann et al., 2018).

Post-fire recruitment failure is also under-studied, poorly quantified (Boucher et al., 2019) and extremely difficult to quantify at a large scale given existing data (see section 4.). In possibly the most extensive study of post-fire recruitment failure in a pine and aspen dominated ecosystem in the US alpine region, Stevens-Rumann et al. (2018) found that shifts in climate had resulted in a significant increase in the rate of recruitment failure since 2000. If similar increases in recruitment failure are occurring throughout the boreal zone, these particular findings may indicate an imminent risk of a widespread state change from boreal forest to a non-forested ecosystem. This increase in the rate of post-fire recruitment failure, combined with both the growing number of studies that have observed it (Barrett et al., 2020), as well as the observed and projected increases in fires frequency and severity (see section 3.1.1) raises the alarming possibility of the ecological collapse of large parts of the boreal ecosystem, leading to the reversal of the boreal carbon sink (Bradshaw and Warkentin, 2015).

2.3 A case study of post-fire recruitment failure in southern Siberia

Southern Siberia is one of the driest and hottest parts of the boreal forest with an average annual precipitation of ~200-300 mm (Figure 1) and is among the fastest warming parts of the boreal zone (See section 3.1). As such, it is expected to be the most vulnerable to climate change and the changes currently observed in this region may be indicative of the future of the boreal zone as the climate warms. This region has already experienced an increase in the length of the fire season and a shortening of the FRI that is expected to continue with climate change (Malevsky-Malevich et al., 2008; Shvetsov et al., 2016) as is considered a hotspot of global forest loss (Achard et al., 2006). A number of recently published studies have looked at long-term post-fire recruitment failure in the southern Eurasian boreal forest range, in the Zabaikalsky Krai and the Republic of Buryatia, immediately southeast of Lake Baikal (Barrett et al., 2020; Kukavskaya et al., 2016; Shvetsov et al., 2019, 2016). In the early 2000s, this region experienced a number of severe fire seasons and in the almost two decades since these fires, it has become apparent that widespread recruitment failure has caused large areas to transition abruptly to grassland ecosystem types (Barrett et al., 2020; Kukavskaya et al., 2016; Shvetsov et al., 2019).

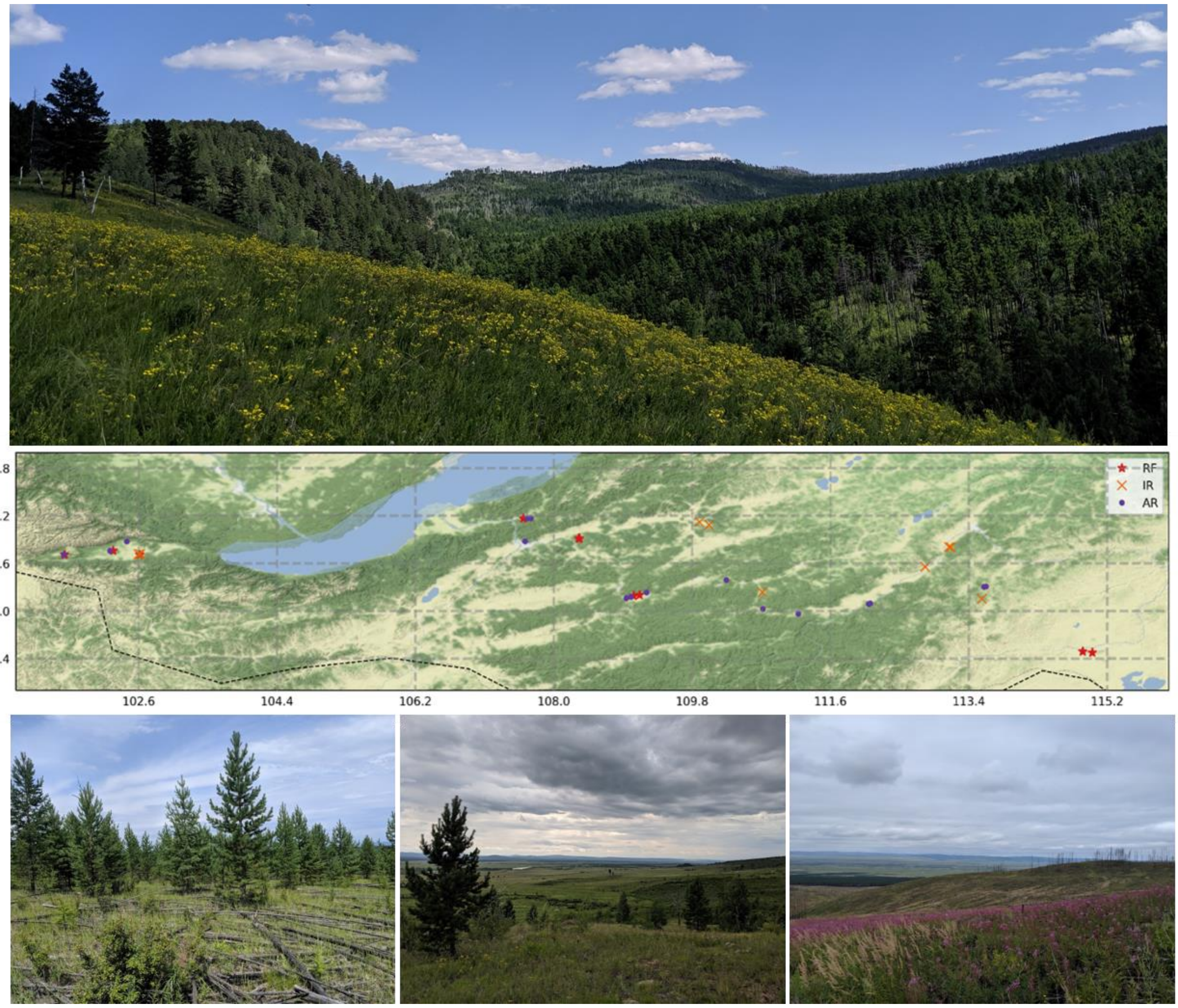

Figure 2 *Photos and map of Siberian field sites. Top: landscape photo showing both the grassland and forest ecosystems present in the study region. Middle: Map of the field sites with recruitment failure (RF), intermediate recruitment (IR) and abundant recruitment (AR) marked. The black dotted line is the Russian-Mongolian border Bottom: Examples of sites with AR (left, IR (middle) and RF (right).*

There is currently no automated method that can be applied to remotely sensed data to identify the different recruitment trajectories in the first 5-10 years after fire, and even 20 years after fire, discriminating recruitment trajectories reliably is difficult (See section 4). Whilst some indicators do show promise, with initial differences in greenness and moisture among sites characterized by abundant recruitment, intermediate recruitment and recruitment failure becoming larger over the 10 years post-fire (Barrett et al., 2020), further work is needed to be able to use them to quantify recruitment trajectories. This means it is currently impossible to do a systematic large-scale assessment of post-fire recruitment failure.

All these recent studies use a combination of field observations to determine the recruitment trajectory and remotely sensed data to determine the impact of factors such as burn severity (Barrett et al., 2020; Kukavskaya et al., 2016; Shvetsov et al., 2019). Barrett et al,. (2020) observed complete recruitment failure in 13 of 64 sites, with a further 37 sites showing less than optimal recruitment (Figure 2). This is nearly double the ~11% recruitment failure observed in the sites examined by Shvetsov et al. (2016). It should be noted that these sites used in all three studies were chosen for accessibility and the ability to clearly see fire impacts in remotely sensed data. As such, they may not be a representative sample of the post-fire recruitment trajectories across the entire region.

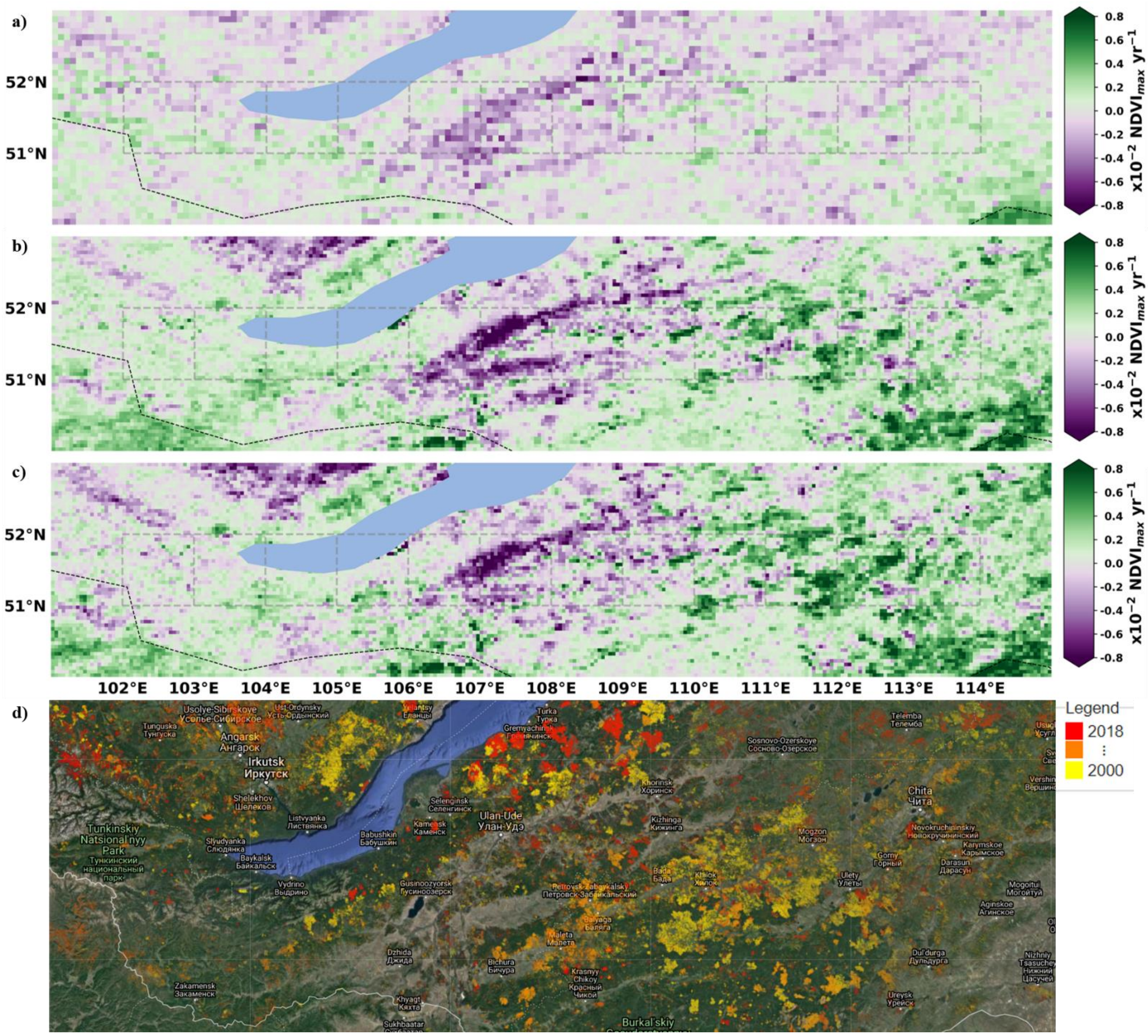

Figure 3 *Maps of the Theil-sen slope in annual $NDVI_{max}$ for a) GIMMS (1982-2017), b) MODIS Terra (2000-2018), MODIS Aqua (2002-2018) and d) Hansen forest loss (2000-2018). The yellow-orange patches are areas of forest loss.*

The regional trend in NDVI, as well as the loss of forest cover since 2000, are shown in Figure 3. Between 1982 and 2017 the region had an average change in $NDVI_{max}$ of -0.0002 ± 0.0015 with 9.2% showing a significant (α_{FDR} = 0.10) positive trend and 13.1% a significant negative trend (α_{FDR} = 0.10). Large hotspots of negative trend can also been seen in the shorter MODIS Terra and MODIS aqua trend (Figure 3b-c). This region has also seen a loss of ~15.9% of the forest cover over the same period (Figure 3d). Whilst these negative trends in NDVI and the widespread forest loss cannot be used to directly quantify the extent of recruitment failure (see section 4.), the fact that they are occurring in the same regions where recruitment failure is being observed in field observations, does provide some degree of corroboration.

Even in areas where post-fire recruitment failure has been identified, determining the exact cause is difficult because vegetation recovery is controlled by complex interactions of multiple factors, including pre- and post-fire climate condition, the FRI, the distance to a seed source, as well as successive disturbances (Liu, 2016; Payette and Delwaide, 2003). All of these factors have been found to play a role in recruitment failure in this region, with multiple fire events being especially predictive of total recruitment failure (Barrett et al., 2020; Kukavskaya et al., 2016; Shvetsov et al., 2019). The importance of these factors has also been observed across the border in northern Mongolia (Otoda et al., 2013).

Barrett et al,. (2020) also found evidence of a possible snow feedback mechanism that would act to reinforce initial differences in recruitment over the intermediate and long term. In the first year after fire, the winter snow coverage as measured by the Normalized Difference Snow Index (NDSI), showed no difference between abundantly recruiting sites and those experiencing recruitment failure (Figure 4). However, 3-4 years after the fire, NDSI values at abundantly recruiting sites were substantially higher than at the recruitment failure sites. Snow cover in winter provides seedlings insulation from cold temperatures (Myers-Smith et al., 2011) plus protection from wind shear and herbivory (Barrett et al., 2020; Myers-Smith et al., 2011; Tape et al., 2010). Snowmelt also acts as an important source of water in spring (Buermann et al., 2018). These results suggest the possibility of a snow-seedling feedback mechanism whereby a greater density of seedlings traps more snow, which in turn protects seedlings from cold temperatures and wind.

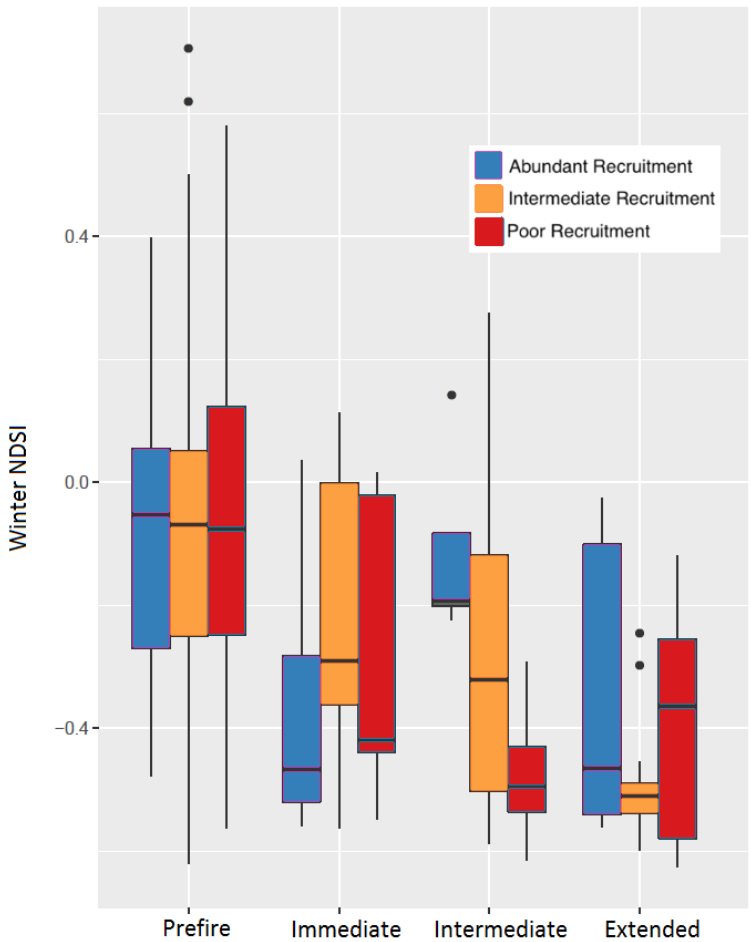

Figure 4 *Winter NDSI values for all assessment periods. Values for AR and RF are significantly different in the intermediate period. Figure adapted from Barrett et al., (2020).*

Whilst the findings of these studies are interesting, they are not definitive and require significant further research for two main reasons. The first is that the field observations took place almost two decades after the stand replacing fires (Barrett et al., 2020). As such, information about factors known to cause recruitment failure such as the fire history and the land management post fire (e.g. salvage logging) is dependent upon the somewhat incomplete satellite record and the memories of individual foresters, if it exists at all. This is especially problematic in the case of salvage logging as successive disturbances are strongly linked to recruitment failure (Kukavskaya et al., 2016, 2016). The practice is widespread in this region (Kukavskaya et al., 2013) and it varies greatly in impact depending upon the exact method used. Post-fire logging is also linked with increased soil erosion, a reduction in the seed bank and a shortening of the FRI (Kukavskaya et al., 2013), which can all cause recruitment failure in their own right.

The second reason further study is needed is that evidence from other regions suggest that multiple drivers are often required for regeneration to fail, which is complicated further by the fact that interactions between drivers are often non-linear and non-independent (Hansen et al., 2018; Payette and Delwaide, 2003). For example, it has been shown that sites adjacent to the edge of a mature forest do better because of much greater seed availability (Kukavskaya et al., 2016, 2016). Forests can also act as wind-breaks which can result it in the deposition of more snow from lowered wind speeds. As such, it is possible, though unlikely, that some, or all, the effect shown in Figure 4 is a secondary impact of distance to forest.

Despite the uncertainty regarding the exact extent and precise combination of the drivers leading to recruitment failure in this region, current evidence from both field studies (Barrett et al., 2020; Kukavskaya et al., 2016; Shvetsov et al., 2019), and remotely-sensed data (Figure 3), suggest that it is widespread. Given that southern Siberia is currently experiencing rapid warming, which will increase in the future, and the growing number of observations of recruitment failure in Europe (Moser et al., 2010), Canada (Payette and Delwaide, 2003) and the United States (Frelich et al., 2017; Hansen et al., 2018; Stevens-Rumann et al., 2018), there is a real possibility that we are seeing the rolling of a regional tipping point leading to the rapid replacement of forest with grasslands with significant consequences for the carbon balance, surface albedo, and the resulting altered energy balance.

3. Drivers of change in the boreal forest zone

The drivers of change in boreal forests can be divided into two broad categories: climate change and land-use; both of which have a range of direct and indirect effects. Though discussed separately, the processes described below are strongly interlinked with a range of positive and negative feedbacks that act to amplify or mitigate the problem.

3.1 Climate change

Over the last 100 years the boreal forest zone has been one of the fastest warming regions in the world (D'Orangeville et al., 2018). The change in temperature and precipitation resulting from climate change over the boreal region is shown in Figure 5a and b respectively. Over the period 1982 to 2017, all the boreal forests experienced significant increases in mean annual temperature, with an average increase of 0.04°C yr^{-1}. Some regions, including northern Canada and southern Siberia, experienced rates as high as 0.06°C yr^{-1}. For comparison, the global mean warming observed over land was 0.03°C yr^{-1}.

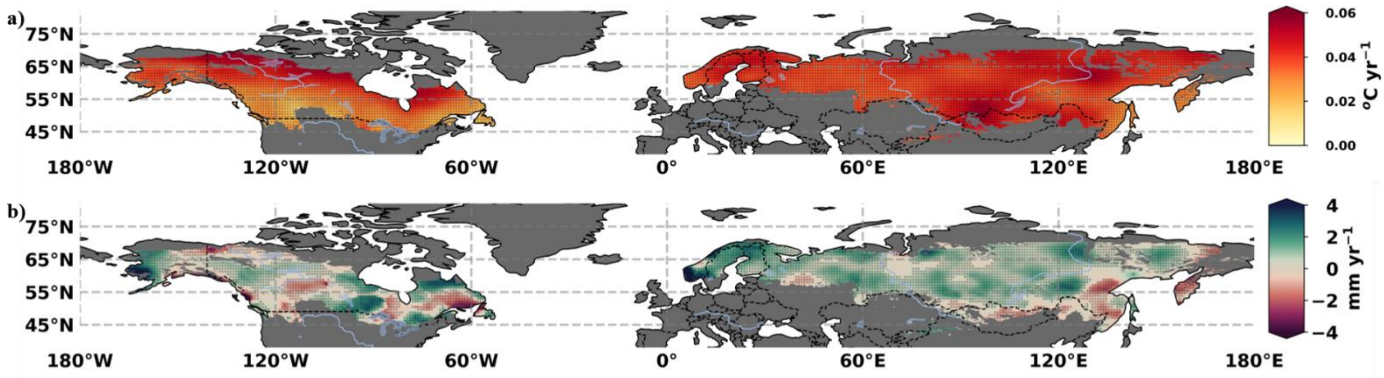

Figure 5 *Trend in a) mean annual temperature for the period 1982-2017 and b) annual precipitation (mm). To control for the impacts of large-scale climate oscillations, a 20 year moving window smoothing was applied and then the trend was estimated using the theil-sen (Theil, 1950). P-values were calculated using a non-parametric Spearman Rho test and then adjusted for multiple comparisons using the Benjamini–Hochberg procedure with a False Discovery Rate of 0.10. Stippling indicates statistical significance and non-boreal forest ecosystems are masked in grey. Data: TerraClimate (Abatzoglou et al., 2018a)*

The global trends in precipitation are more complex than temperature, with 15.9% of boreal forests experiencing significant decreases in rainfall, compared to 64.9% increasing and 19.2% with no significant trend (Figure 5b). Whilst not the largest negative trends, the decreases in rainfall over southern Siberia are of particular importance because they are occurring in an area that already experiences the lowest annual precipitation of the boreal zone (see Figure 1).

3.1.1 Climate Change and increases in the fire regime

In the boreal forest zone, there are strong empirical and conceptual links between burnt area and climate variables such as precipitation and temperature; hotter and drier years generally having larger and more severe fires, with an increase in the frequency of hot dry years linked to shorter FRI's (de Groot et al., 2013; Hanes et al., 2018; Héon et al., 2014; Rogers et al., 2020; Young et al., 2017). Climate change is also linked to increases in the number of lightning strikes, which are the primary source of ignition for many boreal forests (Krawchuk et al., 2009; Veraverbeke et al., 2017). Increases in both the frequency and severity of the fire regime associated with the observed boreal warming have been seen in continental North America (Abatzoglou and Williams, 2016; Turetsky et al., 2015), Canada (Coops et al., 2018; Gillett et al., 2004), Alaska (Beck et al., 2011; Hoecker and Higuera, 2019), China (Liu et al., 2012), Fennoscandia (Aakala et al., 2018) and Siberia (Ponomarev et al., 2016; Stephens et al., 2014). Climate change may also be leading to an increase in the frequency and/or severity of droughts (Pachauri et al., 2014) which are also strongly linked with fires (Aakala et al., 2018; Héon et al., 2014) and with higher fire-induced tree mortality (Ferster et al., 2016).

Increases in fire size and severity, as well as shortening of the FRI, are linked with recruitment failure and forest loss (Hansen et al., 2018; Moser et al., 2010). In the steppe and grassland ecosystems that border boreal forests the FRI is less than 17 years (Frelich et al., 2017), which is less than the ~20 years it takes for the dominant tree species in the drier parts of the boreal forest to reach sexual maturity. In a study of recruitment failure in the alpine region of the continental USA, the regeneration of serotinous lodgepole pine only failed when fire return intervals were <20 yr and stands were far (1 km) from a seed source (Hansen et al., 2018).

3.1.2 Climate change and decreased ecosystem resilience

In addition to increasing the frequency and/or severity of the fire disturbance regime, climate change is also reducing ecosystem resilience (Reich et al., 2018; Stevens-Rumann et al., 2018). Studies have observed an increase in the mortality of mature boreal forest trees, which are the source of seed for post fire recruitment (Hansen et al., 2018). This increased mortality has been linked to direct climate effects such as drought and extreme temperature, as well as indirect effects including increased insect predation (Kharuk et al., 2013) (see (Allen et al., 2010) for a summary of literature by region). Climate can also directly impact post-fire recruitment, with drought in the 1-2 years after a fire decreasing tree recruitment and greatly increasing the likelihood of recruitment failure (Moser et al., 2010; Whitman et al., 2019).

It should be noted that, whilst this section is focused on the challenges climate change presents to boreal forests, there is strong evidence of a greening of large parts of the boreal forest, resulting from the increased temperature in conjunction with CO_2 fertilisation and nitrogen deposition (Chae et al., 2015; Forbes et al., 2010; Keenan and Riley, 2018; Zhu et al., 2016). In the CMIP-5 ensemble results, global greening is projected across the boreal zone for the next century (Piao et al., 2020). However these models have serious limitations in the way they represent vegetation because they cannot capture ecosystem change driven by a range of processes including land-use or changes in the fire regime (Arneth et al., 2017; Bayer et al., 2017; Piao et al., 2020; Zhu et al., 2016). The results of a recent modelling study which used more advanced ecosystem dynamics suggest that large parts of the boreal zone, mostly southern Siberia and central Canada, will see reductions in tree cover by 2050 under all of the IPCC emissions scenarios (Bastin et al., 2019). Ecosystem models have also found similar reductions in regional studies (Mokhov and Chernokulsky, 2010; Stralberg et al., 2018).

3.2 Management and human influence

Nearly two thirds of boreal forests are managed with even unmanged forest being by tourists, fishermen, hunters and beekepers. (Gauthier et al., 2015). After fire, the largest cause of boreal forest disturbance comes from timber harvesting (Brandt et al., 2013; Potapov et al., 2017). In contrast to the deforestation observed in tropical forests, the aim of forest management is a sustainable long-term harvest of timber (Burton et al., 2003). However, forest management reduces the extent of older forests, which can result in lower biological, genetic and structural diversity (Cyr et al., 2009; Gauthier et al., 2015; Melvin et al., 2018). This has the potential to reduce forest resilience to disturbance, especially considering that the ecological impacts of many management strategies remain poorly understood (Melvin et al., 2018; Perrault-Hébert et al., 2017).

3.2.1 Forest Management and the Fire Regime

Whilst there is a widely held belief that logging reduces fuel loads thereby resulting in less severe and less frequent fires (Bradley et al., 2016), studies in boreal regions have found that managed forests experience more fires (Achard et al., 2008; Kukavskaya et al., 2013). This increase has been attributed to two main causes: firstly that logging debris can lead to much higher surface and ground fuel loads (Kukavskaya et al., 2013). Secondly, that humans are the main source of ignition (Achard et al., 2008; Campos-Ruiz et al., 2018; Liu et al., 2012) with more than 80% of fires in some regions being anthropogenic in origin (Brazhnik et al., 2017; Mollicone et al., 2006).

Forest management may also be a factor in post-fire recruitment failure. A common practice in managed forests is to log dead trees after a fire event (Taboada et al., 2018; Thorn et al., 2018). Whilst fires can occur in a forest of any age, structure and composition data (Brazhnik et al., 2017) suggest that the probability of fire is lower in the initial decade after fire (Hart et al., 2019). Post-fire logging offsets this reduced risk and has been linked to increased fire frequency (Donato et al., 2006; Taboada et al., 2018) which has been linked to post-fire recruitment failure (Hansen et al., 2018; Moser et al., 2010). Post-fire logging has been found to hinder the regeneration of forests (Donato et al., 2006), increase soil compaction and erosion (Malvar et al., 2017) and to increase the number of open habitat plant species (Thorn et al., 2018). The negative impact of salvage logging may be especially high in Scots pine ecosystems, such as those found in southern Siberia, as dead standing trees provide a source of seeds in the 1-5 year recruitment window and continue to play a vital role in the ecosystem for up to 200 years after the stand replacing fire occurs (Kuuluvainen et al., 2017). Most studies that assess post-fire logging only look at the first 5 years after the fire event (Boucher et al., 2014; Taboada et al., 2018; Thorn et al., 2018) and further research is needed to examine the link between post-fire recruitment failure and logging after fires.

4. Measuring the scale of the problem

Due to the size of the boreal forest zone, the only way to directly measure large-scale changes is to use remote sensing (Marchand et al., 2018). Remote sensing is being used in two ways: direct disturbance detection (Hansen et al., 2013) and long-term vegetation trend analysis (Marchand et al., 2018). The results have been inconsistent these various approaches, with significant divergence in the estimates of boreal forest gain and loss (Myers-Smith et al., 2020; Piao et al., 2020; Polar Research Board et al., 2019). For example, a recent review of the published studies of trends in productivity of the Canadian boreal forest found that about half the studies identified increasing trends while the other half showed negative trends (Marchand et al., 2018). The authors of that review attributed this discrepancy in the estimates as resulting primarily from differences in the methodology used and the spatial scale of the study.

4.1 Disturbance detection

There have been two approaches to disturbance detection in boreal forests. The first assesses forest loss regardless of the type of disturbance and the second measures parameters specific to fire disturbances. Forest loss detection generally uses Landsat or other medium to high spatial resolution remotely-sensed data along with some form of classification algorithms to measure the extent of forest at specific times, with the differences between observed dates indicating areas of forest loss and gain (Hansen et al., 2013; Potapov et al., 2011, 2008; Schroeder et al., 2011). In ecosystems where there is also large interannual variability in the natural disturbance regime, as is the case with fire in the boreal zone (Beurs et al., 2018), this is especially problematic because different study periods can result in opposing apparent trends.

Even when the data are produced annually, such as the widely used Global Forest Change (GFC) product (Hansen et al., 2013), there are limitations that prevent assessment of the long-term trends in disturbance. The GFC data currently covers the period 2000 to 2018 but, at the time of writing, an improvement to the algorithm implemented in v1.6 to improve the detection of boreal forest loss due to fire, means that the period 2000-2010 is not directly comparable to the results from 2011-2018. In addition to the issues with temporal resolution, there are considerable inconsistencies in the results of different datasets (Li et al., 2017). Despite these limitations, these forest loss datasets offer the best insight into the recent changes in forest extent over the boreal region. In a 2013 study of global forest loss, Hansen et al., (2013) found that boreal forests had both the largest gains of any forest zone and the second largest area of forest lost in both absolute and proportional terms (Figure 6). This finding is consistent with the results of the 2015 Global Forest Resources Assessment performed by the Food and Agriculture Organization of the United Nations (Keenan et al., 2015).

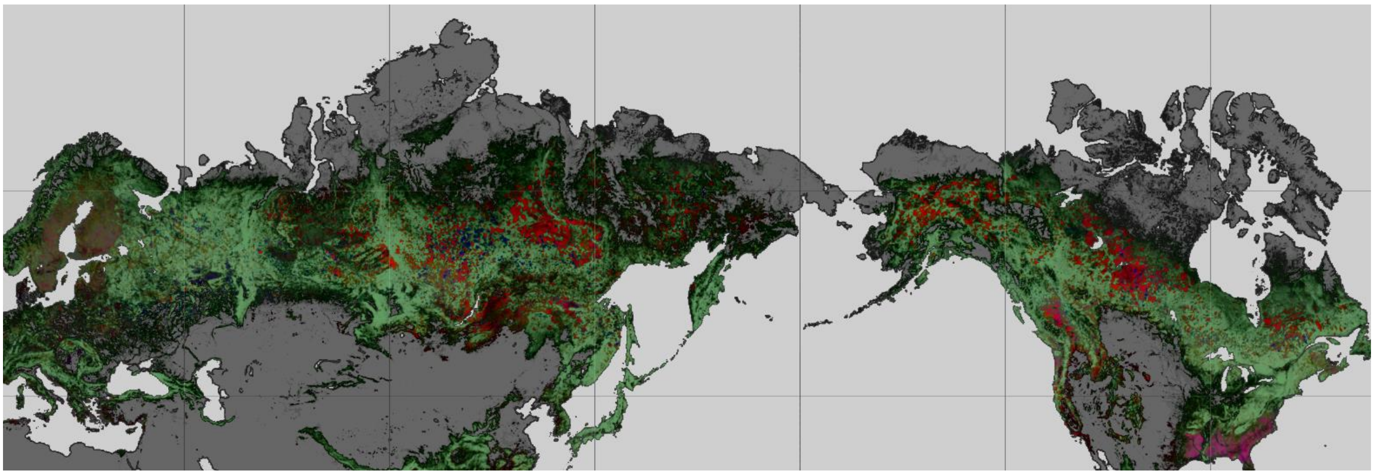

Figure 6 *Global Forest extent (dark green for low tree cover, light green for high tree cover), gain (Blue) (2000–2012), loss (red) (2000-2018) and both loss and gain (purple) determined using the Google Earth Engine and v1.6 of the Hansen et al., (2013) Global forest change dataset.*

The second disturbance detection remote sensing approach focuses on the detection of forest fires. A diverse range of methods have been developed to measure the extent and characteristics of active fires, the area burnt (Humber et al., 2019), the burn severity and the recovery of vegetation after the fire event (Chu and Guo, 2014). The strengths and weakness of these approaches have been addressed in review articles (Chu and Guo, 2014; Mouillot et al., 2014) and dataset comparison studies (Giglio et al., 2010; Humber et al., 2019; Moreno-Ruiz et al., 2019). These articles found that global annual estimates are consistent but there are considerable differences between datasets at regional or biome scales (Humber et al., 2019; Padilla et al., 2014).

Despite the limitations and uncertainties, fire disturbance detection products are more reliable in the boreal zone than in other biomes (Humber et al., 2019; Zhu et al., 2017) with one study finding that MODIS burned area product only under-estimated the extent of the burnt area by 27% compared to a global average of 48% (Padilla et al., 2014). By combining forest loss and fire detection data, recent studies have been able to detect and attribute forest loss (Liu et al., 2019).

4.2 Large scale trends in vegetation

The other way to examine long term changes in boreal forest is to look at trends in vegetation indexes (VI), with the VI is the Normalised Difference Vegetation Index (NDVI) being the most widely used. NDVI is a proxy for Net Primary Production (NPP) (Burrell et al., 2020; Polar Research Board et al., 2019; Prince and Tucker, 1986; Tucker et al., 1985; Wessels et al., 2006; Yang et al., 2017). In the boreal zone, positive changes in NDVI have been linked to field observations of temperature driven increases in NPP whilst browning trends have been linked with fire and climate disturbance (Beck et al., 2011; Myers-Smith et al., 2020; Polar Research Board et al., 2019; Yang et al., 2017).

The main advantage of a trend detection approach is that it is easy to replicate and there are range of datasets available with larger temporal ranges and different resolutions. For example, there are currently four datasets providing global sub-monthly NDVI with more than 17 years of continuous records (GIMMS3.1g, MODIS terra, MODIS aqua and Copernicus). These data sets use the same NDVI formula, enabling researchers to identify and quantify the uncertainties that comes from sensor and scale differences (Burrell et al., 2018), which is difficult with the more direct disturbance quantification approaches. An intercomparison of commonly used NDVI datasets is shown in Figure 7.

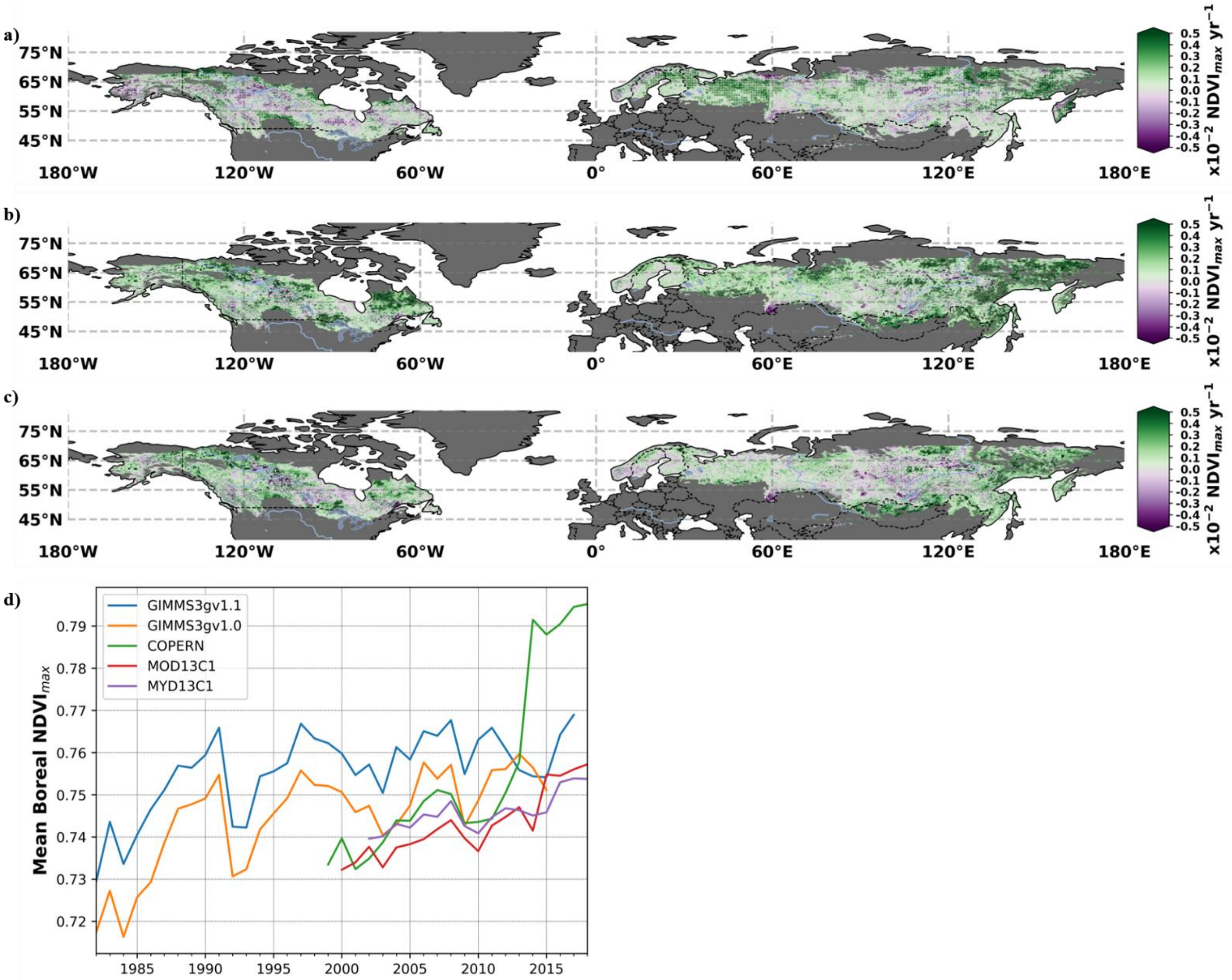

Figure 7 *Slope in Annual Maximum NDVI (1982-2017) determined using the Theil-sen Slope estimator with a) GIMMS3gv1.1 (1982-2017), b)MODIS TERRA (MOD13C1) (2000-2018), c) MODIS AQUA (MYDD13C1)(2002-2018) data. P-values were calculated using a non-parametric Spearman Rho test the adjusted for multiple comparisons using the Benjamini–Hochberg procedure with a False Discovery Rate of 0.10. Stippling indicates statistical significance.*

In general, the NDVI datasets shown in Figure 7 have similar largescale trends, with the boreal zone having greened by an average of 0.00056 ± 0.000003 in the GIMMS dataset (1982-2017), 0.001288 ± 0.000004 in the MODIS TERRA data (2000-2018) and 0.000758 ± 0.000006 in the MODIS Aqua (2002-2018). These findings are consistent with the observed trends in existing studies (Guo et al., 2018; Marchand et al., 2018; Myers-

Smith et al., 2020; Sulla-Menashe et al., 2018). In a study of vegetation trends in the Canadian boreal forest, it was found that greening and browning not caused by the disturbance/recovery regime was primarily located near boundaries of the boreal forest zone, with browning in the dry regions and greening in the wet regions (Sulla-Menashe et al., 2018). Whilst overall vegetation trends are consistent between datasets, regional patterns vary considerably (Figure 7a-c)and this difference can be attributed to differences in the start and end dates, it may also result from errors and uncertainties in the data itself (Burrell et al., 2018). An example of this problem is shown in Figure 7d. The Copernicus NDVI produce is a 1km global NDVI product that covers 1999 to present. In 2013 this dataset changes from SPOT data to the newer high resolution PROBA-V. It is apparent that, for the boreal forest region, the cross-sensor calibration has not worked. All datasets have uncertainties and errors and even small errors can produce contradictory trends in some cases (Burrell et al., 2018). These difference may account for some of the ongoing debate about the arctic and boreal green trends (Duncan et al., 2020; Polar Research Board et al., 2019).

4.3 Detecting post-fire recruitment failure

The threat facing boreal forests is that post-fire recruitment failure will cause large parts of the boreal forest ecosystem to collapse with climate change-driven increases in the fire regime (Walker et al., 2019). Trend analysis alone cannot be used to separate out normal fire loss and recovery from recruitment failure. This is especially problematic because a recent study found that disturbance recovery dynamics account for the majority of NDVI trends in boreal forest (Sulla-Menashe et al., 2018).

This limitation is compounded by the fact that trend detection approaches can be unreliable in any ecosystem where productivity is impacted by natural interannual variability in climate, especially when that variability is linked with decadal time-scale climate oscillation (Burrell et al., 2017). This is the case in the parts of the boreal forest most at risk of collapse due to post fire recruitment failure (Bradshaw and Warkentin, 2015). Similarly, direct disturbance detection methods measure forest loss but not recruitment failure (Hansen et al., 2013) and consequently cannot distinguish natural disturbance recovery cycles from a state change.

In reviewing remote sensing of fire impacts, Chu and Guo (2014) argued that methods for evaluating post-fire vegetation recovery has received "little effort" and that considerable further research is required (Chu and Guo, 2014). The authors of this chapter are aware of no study published to date that has estimated the extent of recruitment failure at a large spatial scale. However, a number of studies have shown promising results by combining disturbance detection and NDVI or other vegetation index trends to monitor post-fire vegetation recovery dynamics (Cai et al., 2018; Chu et al., 2016; Liu, 2016; Yang et al., 2017). In addition, it may be possible to use methods developed in other ecosystems that improve trend detection by accounting for climate variability and ecosystem disturbance (Abel et al., 2019; Burrell et al., 2019) though validiating and improving on these methods remains difficult without largescle gridbased inventories currently lacking in Siberia.

4.4 Post-fire recruitment failure and the prediction of future climate

The boreal forest has a large impact upon regional and global climate (Chen and Loboda, 2018; Helbig et al., 2016; Liu et al., 2019). If post-fire recruitment failure results in the collapse of large parts of the boreal forest it may potentially have a large impact upon global climate that is not currently integrated into many of the models used to predict climate change.

Over the next 100 years, global circulation models (GCM) predict that the boreal region will experience the largest increase in temperatures of any forest biomes (Gauthier et al., 2015). This, along with the associated lengthening of the growing season, is predicted to result in the continued expansion of boreal forests into the tundra region (Forbes et al., 2010; Rocha et al., 2018). GCM's incorporate vegetation in necessarily simplistic ways and there is growing evidence for rapid and non-linear response to changes in climate that are poorly understood, not incorporated into current models and may be leading to significant overestimations of the long-term benefits of warming in the boreal zone (Chen et al., 2016; D'Orangeville et al., 2018; Reich et al., 2018; Soja et al., 2007; Thurner et al., 2017)

One current limit of GCM's and their associated global vegetation models (GVM's) is that, while these models include fire in the carbon cycle, they only consider fire frequency and do not incorporate feedbacks between fire, vegetation, and climate (Harris et al., 2016; Syphard et al., 2018). Consequently, forest loss caused by processes such as post-fire recruitment, cannot be captured. This is problematic because studies predict large increases in fire frequency over the next century (Lehtonen et al., 2016; Wotton et al., 2017, 2010). Given the complex feedbacks that exist between the boreal forest zone and the global climate (Chen and Loboda, 2018; Helbig et al., 2016; Johnstone et al., 2011; Liu et al., 2019; Zhang et al., 2011) widespread collapse of the boreal forest ecosystem due to post-fire recruitment failure may change the regional albedo as well as releasing significant amounts of CO_2 and aerosols into the atmosphere. While some of these issues may be addressed with the development of dynamic global vegetation models (DGVMs) that more accurately represent vegetation processes and fire dynamics (Harris et al., 2016; Syphard et al., 2018), significant further research into forest loss due to post-fire recruitment failure is needed.

5. Future Management

Given that negative impacts of climate change will continue to increase for at least the next century, even under the lowest emissions scenarios, it is necessary to implement management strategies to prevent widespread forest loss or facilitate more desirable transitions. Without knowing the scale of the problem, it is difficult to formulate effective management practices. With increasing fire activity, forests become more vulnerable to changes in species composition and structure (Shvetsov et al., 2019) and monitoring of post-fire recovery is necessary to determine appropriate management approaches. Several forest management and adaptation strategies have been proposed to mitigate the negative impacts of climate change which can be can be grouped into three broad categories: 1) societal adaptation (e.g., forest policy to encourage adaptation, revision of conservation objectives, changes in expectations), 2) adaptation of the forest (e.g., species selection, tree breeding, stand management), and 3) adaptation to the forest (e.g. changes in forestry rotation age, use more salvage wood, modify wood processing technology)(Spittlehouse, 2005).

There is a documented lack of published forest management and restoration research from the Siberian region, with almost all the published literature coming from North America, Finland and Sweden (Bernes et al., 2015).

In southern Siberia, a range of different management strategies are being tested by the regional forestry organizations but they remain mostly unstudied and so their effectiveness is unknown. Most of the strategies employed in this region are focused heavily upon salvage logging immediately after the fire (Kukavskaya et al., 2013) followed by replanting in areas where recruitment failure has been observed. A recent study on global forest regeneration potential found that Russia has about 150 million ha of land suitable for forest restoration, the largest amount of any country (Bastin et al., 2019). By comparison, the annual rate of reforestation in Russia over the last 15 years has been 800-950 thousand ha with forest plantations averaged 22% (http://rosleshoz.gov.ru/). Furthermore, the effectiveness of these measures in reducing the risk of permanent forest loss is dubious due to the usage of low-quality seeds and seedlings, the lack or absence of subsequent measures for protection of seedlings from diseases, insects, and fires as well as a lack of erosion mitigation. Most problematic of all is that ~50% of the areas replanted in the most fire prone parts of Siberia burn again within 15 years (Kukavskaya et al., 2016).

In order to stimulate natural reforestation and to decrease recruitment failure following management intervention, a recent Decree published by the Ministry on Natural Resources and Ecology of the Russian Federation suggested the following measures: preservation of seedlings during harvesting, maintenance of seed sources, enclosing regenerating areas (e.g. cattle fencing), sanitation thinning, seedlings care (e.g., tree setting, fertilizer application, herbicide treatment), mechanical or fire soil mineralization (Decree 188, 2019, p. 139). These measures can be carried out both separately and in combination with each other. For example, to enhance the reforestation effect, thinning of stands and undergrowth, along with increased light input to the crowns (increased seed production) and under the canopy (improved light conditions for regeneration) could be supplemented by soil mineralization. This Decree also provides guidance for forest management on the number of seedlings required for successful regeneration, depending upon the forest zone, dominant tree species, soil type and moisture.

It should be noted that many of the strategies outlined above are already implemented over the extensive areas of Russian boreal forest, despite which, the rate of re-burning and then subsequent recruitment failure remains high (Kukavskaya et al., 2016). As such, the improvement of reforestation management should include strengthening of fire prevention measures, including education of local communities to decrease number of ignitions, as well as construction and maintenance of a fuel break system (prioritizing nearby settlements and tree plantations) to inhibit development of large fires and to decrease the negative impact of fires on forests. Active replanting of forests in western North America following salvage logging of burnt forest may make them more susceptible to re-burning due to high density of saplings. Lindenmayer et al. (2017) therefore suggested reducing the numbers of planted seedlings and increasing the spacing between trees to reduce the risk of recurrent high-severity fire. Other appropriate actions to improve seedling survival after replanting are to modify seed transfer zones and to introduce more fire-resistant and drought-tolerant species, to change replanting methodology and techniques (e.g. shading of planted seedlings on the overheated sites), to conduct sanitation thinning.

Even if all of these strategies are implemented, they may prove ineffective because they are predicated on the assumption that direct management of an ecosystem post-fire is an effective strategy for ecosystem recovery. A recent global multi ecosystem assessment of management interventions in the wake of natural disturbance events like fires found that active management actually worsened long-term outcomes (Lindenmayer et al., 2017). There is good reason to think this may be the case in Siberia, with salvage logging linked to shorter FRI and recruitment failure (Kukavskaya et al., 2013; Shvetsov et al., 2019) and recent field observations finding extensive soil erosion at some recent replanting sites.

In summary, the current a lack of knowledge about the exact scale and mechanisms of boreal forest recruitment failure as well as the effectiveness of measures being implemented to address it means that developing an effective long-term strategy to prevent widespread boreal forest collapse is extremely difficult. There is a need to reconsider our human response to natural disturbances and to reduce the risks of forest degradation by implementing appropriate forest management strategies based upon the local climate and ecological conditions as well as disturbance levels. Without these improvements, there is a risk that the increasing number of disturbed ecosystems and recruitment failure worldwide will lead to irreversible negative consequences affecting both the environment and human wellbeing.

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