Identifying Indicators of State Change and Forecasting Future Vulnerability in Alaskan Boreal Forest

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<thead>
<tr>
<th>Acronym</th>
<th>Full Form</th>
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<tbody>
<tr>
<td>AIC</td>
<td>Akaike information criterion</td>
</tr>
<tr>
<td>AICC</td>
<td>Alaska Interagency Coordination Center</td>
</tr>
<tr>
<td>ALT</td>
<td>Active layer thickness</td>
</tr>
<tr>
<td>ALFRESCO</td>
<td>ALaska FRame-based EcoSystem Code</td>
</tr>
<tr>
<td>ANOVA</td>
<td>Analysis of variance</td>
</tr>
<tr>
<td>C</td>
<td>Carbon</td>
</tr>
<tr>
<td>Ca</td>
<td>Calcium</td>
</tr>
<tr>
<td>CCMA</td>
<td>Canadian Centre for Climate Modeling and Analysis</td>
</tr>
<tr>
<td>CMIP3</td>
<td>Coupled Model Intercomparison Project Phase 3</td>
</tr>
<tr>
<td>CMIP5</td>
<td>Coupled Model Intercomparison Project Phase 5</td>
</tr>
<tr>
<td>CO2</td>
<td>Carbon dioxide</td>
</tr>
<tr>
<td>CPCRW</td>
<td>Caribou-Poker Creeks Research Watershed</td>
</tr>
<tr>
<td>DBH</td>
<td>Diameter at breast height</td>
</tr>
<tr>
<td>DOS-TEM</td>
<td>Dynamic organic soil-Terrestrial Ecosystem Model</td>
</tr>
<tr>
<td>ET</td>
<td>Evapotranspiration</td>
</tr>
<tr>
<td>FRI</td>
<td>Fire return interval</td>
</tr>
<tr>
<td>GCM</td>
<td>Global Climate Model</td>
</tr>
<tr>
<td>GPS</td>
<td>Global positioning system</td>
</tr>
<tr>
<td>IEM</td>
<td>Integrated Ecosystem Model</td>
</tr>
<tr>
<td>K</td>
<td>Potassium</td>
</tr>
<tr>
<td>KCl</td>
<td>Potassium chloride</td>
</tr>
<tr>
<td>LTER</td>
<td>Long Term Ecological Research</td>
</tr>
<tr>
<td>MANOVA</td>
<td>Multivariate analysis of variance</td>
</tr>
<tr>
<td>Mg</td>
<td>Magnesium</td>
</tr>
<tr>
<td>N</td>
<td>Nitrogen</td>
</tr>
<tr>
<td>NH₄</td>
<td>Ammonium</td>
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<tr>
<td>NH₄Cl</td>
<td>Ammonium chloride</td>
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<tr>
<td>NO₃</td>
<td>Nitrate</td>
</tr>
<tr>
<td>OL</td>
<td>Organic layer</td>
</tr>
<tr>
<td>P</td>
<td>Phosphorus</td>
</tr>
<tr>
<td>PET</td>
<td>Potential evapotranspiration</td>
</tr>
<tr>
<td>PFRR</td>
<td>Poker Flats Research Range</td>
</tr>
<tr>
<td>PLS</td>
<td>Partial least squares</td>
</tr>
<tr>
<td>PM</td>
<td>Picea mariana</td>
</tr>
<tr>
<td>PRISM</td>
<td>Parameter-elevation Regressions on Independent Slopes Model</td>
</tr>
<tr>
<td>PVC</td>
<td>Polyvinyl chloride</td>
</tr>
<tr>
<td>QA/QC</td>
<td>Quality assurance/Quality control</td>
</tr>
<tr>
<td>ROL</td>
<td>Relative loss of organic layer</td>
</tr>
<tr>
<td>SNAP</td>
<td>Scenarios Network for Alaska and Arctic Planning</td>
</tr>
</tbody>
</table>
TEM Terrestrial Ecosystem model
USGS US Geological Survey
Keywords

Alaska paper birch, base cations, Betula neoalaskana, black spruce, boreal forest, carbon, climate change, dendroclimatology, drought stress, ecological modeling, feedbacks, fire, fire effects, fire management, fire severity, forest regeneration, growth-climate responses, invasive species, model coupling, nitrogen, nutrient cycling, permafrost, Picea mariana, plant-soil-microbial feedbacks, radiocarbon, stable carbon isotopes, seedling establishment, succession, soil, succession, tree-rings, tree species, vegetation dynamics

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Objective

This research is designed to understand the mechanistic connections among vegetation, the organic soil layer, and permafrost ground stability in Alaskan boreal ecosystems. Understanding these linkages is critical for projecting the impact of climate change on permafrost in ecosystems that are subject to abrupt anthropogenic and natural disturbances (fire) to the organic layer. We hypothesize that major threshold change is more likely to occur in ecosystems that are already at the margins - forests that, historically, are already stressed - and in fires that are at the extremes in terms of size or severity. We expect that severe fires occurring in forest stands that have not experienced deep burning as part of the recent fire cycle will consume a larger proportion of the organic soil layer and have the greatest potential for permafrost destabilization. We hypothesize that drought-stressed forest stands are more likely to shift to an alternate, deciduous, successional trajectory after fire and that moss percent cover and organic soil re-accumulation are negatively related to the percent cover of deciduous canopy tree species.

This research responds to several elements within the SERDP Statement of Need by using Department of Defense (DoD) and surrounding lands in Interior Alaska to explore and test the conceptual and mechanistic basis for threshold change and regime shift in the Alaskan boreal forest (SON 4). In this interim report, we focus largely on Objective 1 of our original proposal to: determine mechanistic links among fire, soils, permafrost, and vegetation succession in order to develop and test field-based ecosystem indicators that can be used to directly predict ecosystem vulnerability to state change. This objective has been addressed through extensive field-based measurements that have refined our understanding of the linkages between vegetation, organic soils, and permafrost and when threshold changes are likely. We are now focused on incorporating this field data into spatially explicit numerical modeling to predict the response of permafrost ground and disturbance regimes to projected changes in climate (SON 2). These modeling approaches will provide a dynamic mapping tool to help land managers identify those DoD lands that are resistant and those that are vulnerable to permafrost degradation. The modeling will also explore the consequences of interactive changes in climate and management for vegetation composition, establishment of invasive species, fire dynamics, and ecosystem structure and function (SON 1, 3).
Technical Approach

Task 1: Field sampling of wildfire site network

In summer 2011, our field crew resampled the black spruce wildfire network that was established 6 years ago following the 2004 severe fire year in Alaska. The study area consisted of three large burn complexes located in interior Alaska. The burns originated from multiple fire starts that occurred during the hot and dry summer of 2004. In the year after the fire, we selected 38 stands that were dominated by black spruce prior to burning for detailed experiments on post-fire regeneration. Measurements of environmental characteristics, fire severity (Boby et al. 2010), post-fire seed rain (Johnstone et al. 2009), and initial plant recovery were made at the sites in 2005-2008 (Johnstone et al. 2010, Bernhardt et al. 2010). Because these sites span a wide range of soil moisture (from sub-xeric to sub-hygric), elevation (lowland valley forests to alpine treeline) and fire severity (~0 to ~100% soil organic layer (OL) consumption), they provide an excellent suite of sites to test and quantify the effects of fire severity on plant and soil processes across broad landscape gradients. Our initial measurements indicated that severe burning at many sites has interrupted the feedbacks that favor black spruce stand replacement, and stimulated the initiation of alternative successional trajectories via increased recruitment of deciduous tree species (Johnstone et al. 2010). However, it remains unclear whether the patterns of initial seedling recruitment are likely to be maintained, augmented, or diluted by longer-term effects of site conditions and fire severity on soil thermal regime, nutrient pool sizes, rates of nutrient cycling, and multi-year growth and survival of tree seedlings after fire.

Sampling activities for this SERDP project built off the previous records of environmental conditions at the sites and included re-measurements of soil OL and active layer depth along permanent monitoring transects, assessment of tree seedling recruitment, growth, and nutrient status, as well as harvest of an outplanting experiment to test for controls over establishment of dominant tree species (Task 1.1).

Soil OL depth and depth to permafrost were re-measured at 5 m intervals along two 25 m transects at each site. To measure OL depth, a knife was used to remove a ~10 cm x 10 cm intact square of organic material, then a ruler was used to measure the depth of each horizon. Depth to permafrost was estimated at a location adjacent to the organic soil measurement by inserting a 2 m steel probe into the ground and recording the depth at which frozen soil was encountered.

In 2011-2012, we processed ~10,000 seedlings from the outplanting experiment in the lab, separating and weighing biomass in categories of current year and previous year tissues to obtain estimates of annual and total growth (Task 1.2). Foliage samples from harvested seedlings of 5 tree species were ground to a fine powder and prepared for nutrient analyses. Data on seedling biomass were assembled and catalogued for all years of available data (2005-2011) (Task 1.3). We developed allometric equations to predict biomass of all unharvested individuals for each measurement year. Quality checking was done by plotting the ~43,000 raw data points as well as model residuals for each species and year individually. These outliers were verified against the original field notes and raw data records. If a mistake was found, it was corrected and noted in the data file. Data points with irreconcilable errors were removed and noted (< 1% of total data points); some extreme values were removed because they represented dieback, and these are also
noted in the data files. Allometric equations were developed using log transformed data and (x+c) for new growth to deal with zero values, where c is the minimum non-zero value observed. Linear models using ‘lm’ function in the base package of R statistical software were created using transformed data, and the best fit were selected based on AIC, F statistic, and residual standard error values (Tables 1 and 2). For 2006 and 2008 biomass estimations, we used allometric equations containing only diameter measurements; however for 2011 biomass we used allometric equations that included both diameter and height. This was because new growth height was measured in 2006 and 2008, and total height was measured in 2011.

Using our model coefficients and fitted values, we developed allometric equations to predict current year and total aboveground biomass (g dry mass) for each species in every measurement year. Biomass predictions are based on field measurements of stem diameter and height, and allow us to estimate seedling growth from the initial measurements in 2006 to the harvest in 2011. These equations are of high accuracy, with R² values generally above 0.8 (Tables 1.1 and 1.2).

In February 2014 we will begin our analyses of data on naturally recruited seedlings by processing the 2011 harvest data for each species and calculating mean stem weight (g aboveground dry mass per individual). We will use field measurements to calculate mean stem diameter for each plot and estimate allometric relationships using mean stem diameter and mean stem weight (g aboveground dry mass). We will use the same process to QA/QC our sown seedling data as we used for the transplanted seedling data, checking for outliers against the raw data files. Our analyses will use R to generate linear models with the best fit will be used for allometric equations so we can estimate the biomass for non-harvested plots based on field measurements of stem diameter for each species sown. Once we have assembled biomass and survival records for all planted and natural seedlings observed in our sample plots, we will proceed with the final steps in our analyses, which are to develop statistical models relating tree growth and survival to environmental, pre-fire stand, and post-fire characteristics.
Table 1.1 Model fit summary table for total biomass using 2011 harvest data. SE = standard error, DF = degrees of freedom.

<table>
<thead>
<tr>
<th>Species</th>
<th>Model covariates</th>
<th>Adjusted $R^2$</th>
<th>P value</th>
<th>Residual SE</th>
<th>F statistic</th>
<th>DF</th>
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</thead>
<tbody>
<tr>
<td>Paper birch</td>
<td>diameter*height</td>
<td>0.9617</td>
<td>&lt;0.001</td>
<td>0.1385</td>
<td>394.7</td>
<td>44</td>
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<tr>
<td>Paper birch</td>
<td>diameter</td>
<td>0.9535</td>
<td>&lt;0.001</td>
<td>0.1705</td>
<td>926.2</td>
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<td>Black spruce</td>
<td>diameter*height</td>
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<td>0.1431</td>
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<tr>
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<td>0.8974</td>
<td>&lt;0.001</td>
<td>0.1805</td>
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<td>0.123</td>
<td>3565</td>
<td>211</td>
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<tr>
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<td>0.1514</td>
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<td>0.8335</td>
<td>&lt;0.001</td>
<td>0.2617</td>
<td>1132</td>
<td>225</td>
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</tbody>
</table>
Table 1.2. Model fit summary table for new (current year) biomass using 2011 harvest data. SE = standard error, DF = degrees of freedom.

<table>
<thead>
<tr>
<th>Species</th>
<th>Model covariates</th>
<th>Adjusted R²</th>
<th>P value</th>
<th>Residual SE</th>
<th>F statistic</th>
<th>DF</th>
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<tr>
<td>Paper birch</td>
<td>diameter*height</td>
<td>0.9441</td>
<td>&lt;0.001</td>
<td>0.1655</td>
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<td>Paper birch</td>
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<td>&lt;0.001</td>
<td>0.181</td>
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<tr>
<td>Black spruce</td>
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<td>0.2052</td>
<td>1168</td>
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<td>0.2166</td>
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<td>0.1934</td>
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<td>0.3015</td>
<td>772.3</td>
<td>225</td>
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</table>
Task 2: Identification and sampling of new wildfire plots and unburned control plots

To extend our understanding of the impacts of wildfire on the soil OL, permafrost, and vegetation beyond recently burned black spruce stands, we identified burned/unburned pairs of sites composed of either a mixture of deciduous and conifer species or dominated by deciduous hardwoods. These sites were used for estimation of age of soil OL material consumed during fire, and instrumented with sensors to monitor long-term change in permafrost.

In 2011, soil OL samples were collected from 7 burned black spruce stands varying in slope, aspect, and drainage class, to quantify carbon (C) losses following fire across different landscape positions and fire severity (Task 2.1). Estimates of OL depth were also collected in adjacent unburned stands. At each site, one 30 m transect was established and OL depth by horizon was recorded every 3 m. At 6 m intervals, the OL was removed in a ~10 cm x 10 cm intact block and returned to the lab for analysis (Task 2.2) of the $^{14}$C in decomposing moss bodies buried in the OL (methods detailed in Mack et al. 2011). The $^{14}$C signature of the moss is an indicator of the $^{14}$C composition of the atmosphere at the time the moss was alive and fixing C. This information allows us to estimate the age of buried moss and in areas that have recently burned tells us how old the organic material was that was consumed during the fire.

Also in 2011, we identified and sampled fire severity, combustion losses, and tree seedling establishment in 12 burned/unburned pairs of mixed and hardwood dominated stands. At each site, one 100 m transect was established. All trees rooted within 1 m of either side of the transect were measured over ten, 10-m intervals for the length of the transect (totaling 100 m$^2$ sampled per transect). All trees > 1.4 m in height were measured at diameter at breast height (DBH, 1.4 m). For trees < 1.4 m tall, a basal diameter measurement was taken. Tree species and percent combustion of aboveground biomass were also recorded. Soil OL depth was measured using a knife to remove a ~10 cm x 10 cm intact square of organic material every 5 m along the transect and a ruler was used to estimate depths of each horizon.

To monitor changes in permafrost following fire, we identified 3 burned/unburned pairs of black spruce sites in 2012 and 2013 (Task 2.3) and instrumented each site with thermistor temperature sensors to a depth of 1.5 m. These sites included forests burned in 2004, 2010, and 2013, two of which are located in fire scars extending onto military lands at Eielson Air Force Base and Fort Wainwright. At each site, a 1.5 m drill bit was used to drill a hole in the soil. A 1.5 m probe containing the temperature sensors was then inserted into the hole so that the top of the probe was level with the soil OL surface. The thermistor wires were inserted into a Campbell Scientific multiplexer and CR1000 datalogger and the datalogger was attached to a battery and solar panel power supply. This automated system records the temperature of each sensor every hour and will provide us with a continuous measurement of differences in permafrost between burned and adjacent unburned forest (results forthcoming).
Task 3: Identification and sampling of fire management plots

Fire management in the boreal forest has become a common practice to reduce fire spread and lessen fire impacts on human and infrastructure safety. To investigate how these practices impact soils, permafrost, and vegetation, we identified and sampled 16 shearbladed and 10 thinned areas located on military, state, and federally managed lands in Interior Alaska. Sites were sampled between 2011-2013 (Tasks 3.1 and 3.3) and included areas at Fort Wainwright, Fort Greeley, and Eielson Air Force Base. At each site, two 20 m transects located ~20 m apart were established in managed and adjacent unmanaged black spruce forest. All trees located within 1 m of either side of the transect line were identified by species and measured at DBH height (when > 1.4 m tall) or basal diameter (height < 1.4 m). Coarse woody debris was measured using the line intercept method (Brown 1974). Vegetation cover was assessed using the point intercept method (Goodall 1952) every 1 m along the length of the transect. Seedling density was estimated by counting all seedlings present and identifying species within a 1 m x 1 m quadrat placed at 5 random locations along the transect. In the same 5 quadrats, soil OL depth was estimated by using a knife to remove a ~10 cm x 10 cm block of organic material. Each organic horizon depth was measured using a ruler. At 2 of the 5 locations the OL was collected for laboratory analysis. The top 10 cm of mineral soil were also collected in these locations. Samples were collected in the field, put in a cooler, and then frozen until analysis. Thaw depth was estimated adjacent to each soil sampling location using a 1.5 m steel probe that was inserted into the ground to the depth at which frozen soil was hit.

Aboveground live tree biomass estimates were made using published allometric equations (Alexander et al. 2012) and the DBH and basal diameter measurements collected in the field. Soil samples were processed (Task 3.2) by first removing them from the freezer and allowing them to thaw. Vascular plant material was removed from the surface of all OL cores. Next, the sample was weighed and dimensions were recorded. The core was then split in half vertically using an electric turkey cutter. One half was wrapped and returned to the freezer and the other was prepared for additional analysis. The weight and dimensions of the half to be analyzed were recorded and the green moss was removed. Roots > 2 mm in diameter and large pieces of non-decayed wood were removed and the remaining material was thoroughly homogenized using pruners. All roots and green moss samples were weighed wet, dried at 60°C, then weighed again. A subsample of the homogenized OL was weighed wet and also dried at 60°C to estimate moisture content. Mineral soils were split into 0-5 cm and 5-10 cm increments and weighed. Each increment was then pulled apart by hand and rocks and > 2 mm roots were removed. Roots were weighed wet, dried at 60°C, and weighed again. Rock volume was estimated by water displacement in a graduated cylinder. The mineral soil was homogenized and any fine roots were chopped up with pruners. Next, a subsample was weighed and dried at 110°C to estimate moisture content. An additional subsample was dried at 60°C for C and nitrogen (N) analysis. The dried green moss and homogenized organic material were ground in a wiley mill (Thomas Scientific model 3383-L10) or coffee grinder and mineral soil was ground with a mortar and pestle. The ground samples were then analyzed for percent C and N using a Costech Analytical ECS 4010 Elemental Analyzer (Valencia, CA). Soil pH was measured on all OL and mineral soils using a Thermo Scientific Orion 2 Star pH meter. For organic soils, 5 grams of material were mixed with 50 mL of deionized water in a cup and for mineral soils 20 mL of water were
added. Each sample was well mixed, allowed to equilibrate for 30 minutes, then the pH was recorded once the value stabilized.

**Task 4: Moss and litter manipulation experiment**

**Site information**

In summer 2012, we established 30 intensive study plots in the Tanana Valley State Forest, near Fairbanks Alaska (64° 53' N, 148° 23' W) to assess differences in plant-soil-microbial feedbacks between forests dominated by Alaska paper birch (*Betula neoalaskana*) and black spruce (*Picea mariana*) (Task 4.1, 4.3). The forest in this area established following the Murphy Dome fire in 1958 and allowed us to quantify C and nutrient cycling in mid-successional forest stands and to investigate the impacts of deciduous foliar litter on moss growth and survival. Three spatially explicit blocks containing adjacent stands dominated by Alaska paper birch and black spruce were identified and 5 plots (10 m x 10 m) were established for each species in each block, for a total of 15 plots dominated by each species.

**Field Methods**

**Environmental measurements**

In June 2012, we installed 1 meterological station (Onset, Bourne MA) in each stand type within each block (n = 6). At each station air temperature, relative humidity, photosynthetically active radiation, soil temperature (10 cm from the OL surface) and soil moisture (10 cm from the mineral soil surface) are being measured every hour. Two additional soil temperature sensors (iButton®) were installed at 10 cm below the OL surface in each plot to provide additional records of soil temperature every 4 hours. Data are downloaded 2 times per year. To estimate depth of the snow pack, iButton® temperature sensors were installed on 1 wooden stake located near the meteorological station in each stand in September 2012 (Lewkowicz 2008). Sensors were placed on the stake at heights 2, 16, 29, 42, 55, 68, 81, 94, 107, and 120 cm and record the temperature every 4 hours. Data were downloaded in June 2013 and sensors were re-deployed in August 2013.

**Aboveground and understory**

To quantify aboveground biomass and C pools, we recorded diameter and species of all live trees and standing dead trees within each plot. Trees taller than 1.4m were measured at DBH height and basal diameter measurements were taken on trees < 1.4 m in height. Dead trees that had fallen at an angle of < 45 degrees were considered part of the downed woody debris pool while those fallen at an angle > 45 degrees were categorized as a standing dead tree, with DBH or basal diameter measurements recorded. Downed woody debris was quantified using the line intercept method (Brown 1974). Three 10m transects were placed at random locations across each plot and the number of intercepts of each of 5 size classes were recorded. The forest floor was characterized using the point frame method (Schuur et al. 2007). At 10 random locations within each plot, a 0.5 m x 0.5 m point frame was placed on the OL surface. Fishing line was strung at
10cm intervals across the frame, creating 16 intersections within the grid. At each intersection, a pin flag was used to quantify hits on moss, herbs, lichen, shrubs, and “other” plants.

Fresh foliage from the upper canopy was collected from each plot in July 2012. In black spruce stands, tall pruners were used to retrieve samples and in birch stands a slingshot was used. For the birch, 10 intact leaves that were representative of the collection were selected for specific leaf area (SLA) analysis. For the spruce, ~100 needles from the current year’s growth were removed from the branches using scissors or a razor. Samples were shipped on ice to Florida for SLA analysis. Black spruce needles on the tree that had been produced in previous years were also retained and used for chemical analyses (details below).

Annual litter production was assessed using litter collection baskets within each plot. In July 2012, three litter baskets (62.2 cm x 45.4 cm) lined with fiberglass window screen were installed at random locations within each birch plot. In the spruce stands, 50 cm x 50 cm wooden frames with fiberglass lining the bottom were installed. These wooden baskets sit close to the ground and are able to capture needles falling from low limbs on the spruce trees. Litter was collected from birch baskets following leaf fall in September 2012 and October 2013 and from all baskets in June and July 2013.

Moss transplant and deciduous litter manipulation

To assess the impact of deciduous leaf litter on moss growth (Task 4.2), we installed a moss transplant and litter addition experiment across our study plots. *Hylocomium splendens* was selected as the target moss species because, as its common name “stair-step moss” suggests, it forms a new ‘step’ (hereafter called shoot) every year (Figure 4.1). This modular growth and strong apical dominance facilitates monitoring and measurement of growth (Tamm 1953, Busby et al. 1978). Moreover, *H. splendens* is very abundant in the circumpolar boreal forest, as well as in interior Alaska (Økland 1995, Turetsky 2003). The production of new shoots (steps) occurs by branching, i.e. the new shoot grows on the shoot of the previous year. It takes about 1.5 years for a shoot to be considered mature and to have reached its full growth. After about 3-4 years, the shoots become buried, photosynthesis stops, and decomposition begins (Økland 1995).

Within each of the 15 black spruce plots, we identified 6 areas with a high abundance of *H. splendens*. Next, we excavated plugs of live moss measuring 30.48 cm in diameter down to the fibric horizon of the OL in each of the 6 areas. Once the 30 moss plugs from one black spruce stand were collected, they were randomly reassigned to either birch or spruce stands in the same block. In the spruce stands, we randomly chose 3 out of the 6 harvested locations to be replaced with randomized transplanted plugs. In the birch stands, we removed the forest floor to the mineral soil using the 30.48 cm frame and inserted the moss transplant into the hole. We delineated moss transplants using bamboo sticks. All mosses were randomly assigned to one of 3 treatments: litter exclusion, litter addition, and ambient litter (Figure 4.2). In August of 2012 and 2013, we installed plastic mesh tents over the litter addition and exclusion transplants to prevent deciduous leaf litter from falling on them. The mesh contained holes that were large enough to allow precipitation and light to reach the transplant, but prevented the deciduous litter from accumulating. In October 2012 and 2013, we added birch litter manually to the litter addition transplants at the ambient input rate (estimated from the litter collection baskets outlined above). The added birch foliar litter was a well-homogenized sample comprised of litter collected from all the birch litter baskets across the site. In the ambient litter treatment, the transplants received
Figure 4.1 Measurements method for the growth of *H. splendens*. The red arrows represent the measurements on mature previous year shoots, and green arrows represent measurements on current year growth shoots A) PVC markers on a shoot. B) Length measurements. C) Width measurements (modified from Økland (1995)).

Figure 4.2 Leaf litter treatments on moss transplants. Transplants in black spruce stands are shown in the first row, and those in Alaskan paper birch stands are shown in the second row. Treatments are shown in the columns (1 - ambient litter, 2 - litter exclusion, 3 - litter addition).
the natural litter input from the stand they are located in. We identified control areas with a high abundance of *H. splendens* in each plot. In some of the birch stands, these areas were small or found on decomposing logs. In 2013, we installed iButtons® under each transplant to record the soil temperature.

**Moss growth measurements and species composition**

We marked five *H. splendens* shoots in each of the 120 transplants in June 2013, and an additional 5 shoots in September 2013 (Task 4.4). We used PVC rings (HAMA plastic beads, Malte Haaning Plastics Co., Nykøbing Mors, Denmark; outer diameter 2.5 mm and inner diameter 1 mm) with a slit that allowed the rings to be placed as markers on the shoots using pincers (Figure 4.1). The marked *H. splendens* shoots were chosen systematically using a grid system. We measured width and length of the current year’s growth and the new year’s growth on the marked mosses in June and September 2013 (Figure 4.2). We will assess mortality of the mosses in the transplants in a qualitative way. Brown and dry shoots that do not produce new shoots in the fall or the spring will be considered as dead after 2 consecutive years. Mortality will be recorded on the 10 marked shoots.

To see if the moss species composition in the transplants changes following the litter treatments, we used point-intercept sampling (Goodall 1952) with a 100 points grid of which ~76 points fall in the transplant. The grid was positioned in the same place over the transplant for assessments in September 2012 and October 2013. We recorded every moss species that touched a metal pin flag pushed down at every point. *H. splendens* records were visually called green or brown to have a qualitative idea of their condition. Leaf litter cover on the transplants was calculated using perpendicular pictures of the transplants taken in October 2012, June 2013, and October 2013. Pictures were processed to obtain the percentage of leaf cover on the transplants using Adobe Photoshop and ImageJ (Rasband 2012).

At the end of the 2013 growing season (October 2013), we collected one 10 cm diameter core of *H. splendens* in the proximity of each plot. These samples were dried and brought back to the laboratory at the University of Saskatchewan. Measurement of the shoot density, as well as length, width and weight of individual shoots will be assessed in the laboratory during 2014. We will use those results to build allometric equations for biomass accumulation of *H. splendens* in each stand as defined by Økland (1995) and by Benscoter and Vitt (2007).

**Soil**

The soil OL and top 10 cm of mineral soil were collected from 3 random locations within each study plot in August 2012. A ~10 cm x 10 cm intact block of the OL was first removed using a knife and pruners. The total OL depth and depths of green moss, brown moss, and fibric horizons was recorded. The mineral soil immediately below the OL was removed using a 6.8 m diameter soil corer. Samples were wrapped, put on ice, then frozen until analysis.

To estimate N and phosphorus (P) pools in the field, 1 anion and 1 cation resin bag were deployed in the OL and mineral soil within each plot in August 2012. The OL bags were placed halfway between the OL surface and top of the mineral soil. This was accomplished by cutting 3 sides of a 10 cm x 10 cm square with a knife, gently flipping up the OL, and then cutting a horizontal insert into the side of the hole. Mineral soil resins were installed ~ 5 cm below the
surface of the mineral soil. An OL square near (but not obstructing) the OL resin bags was removed, then a trowel was inserted into the mineral soil at one of the edges of the hole, so that the resin bag was located under undisturbed OL. The removed OL square was then placed back into the hole. All birch resins and the OL spruce resins were removed and replaced with new bags in June 2013. In August 2013, all bags were removed and a new set was installed. All resins were rinsed with deionized water after returning to the lab and refrigerated until analysis (see below).

A 5-year mass-loss foliar litter and wood decomposition experiment was installed in August 2012. Senesced black spruce needles, senesced birch leaves, and brown moss had been placed into mesh bags prior to being taken to the field (details below). Birch craft sticks were also deployed as a uniform wood sample. Bags were grouped into sets that contained 1 bag of each litter type and 1 craft stick. Bags within a set were tied together with fishing line, leaving at least 10 cm between bags. 5 sets of bags were then deployed in each study plot around a central stake and fanned out in a star pattern. An additional craft stick was also deployed for removal after the litter portion of the experiment. The first set of bags were retrieved from each plot in August 2013 and frozen until analysis (pending).

**Laboratory Methods**

*Foliage and litter*

Specific leaf area was estimated on a subsample of fresh foliage produced in 2012. All remaining foliage was dried at 60°C following field collection. Leaf litter obtained from birch plots in fall 2012 was sorted immediately into birch foliar litter and other components. A subsample from each plot (composite of the 3 baskets) was then dried at 60°C to estimate moisture and for chemical analyses. Non-foliar litter components and foliar litter collected in summer 2013 from all plots was dried at 60°C following each field collection, then sorted into foliar litter (by species), wood, seeds, cones, and other miscellaneous components. After sorting, all samples were re-dried and weighed. Spruce foliar litter collected in July 2013 from the 3 baskets in each spruce plot were composited into one plot sample for chemical analysis. For both species, a subsample of dried live foliage (both current and past year’s needles for spruce) and foliar litter were ground using a wiley mill (Thomas Scientific model 3383-L10). A portion of this material was analyzed for percent C and N using a Costech Analytical ECS 4010 Elemental Analyzer (Valencia, CA) and an additional subsample was shipped to the Louisiana State University AgCenter for total P, calcium (Ca), magnesium (Mg), and potassium (K) analysis. Briefly, 5 mL of concentrated nitric acid was added to 0.5 g of ground sample. After 50 minutes, 3 mL of hydrogen peroxide was added and the sample was left to digest for 2.75 hours on a heat block. Sample was then cooled and diluted, then run on a Spectro ARCOS iCAP inductively coupled plasma spectrometer (Germany).

*Soil*

Samples were removed from the freezer and allowed to thaw prior to processing. Vascular plants and green moss were removed from the top of the OL samples and dimensions of each organic horizon were recorded. Green moss was weighed, then dried at 60°C. For spruce samples, the
remaining OL was cut horizontally at the interface between the brown moss and fibric horizons, when a brown moss layer was present. Birch organic samples consisted of a partially decomposed litter layer and fibric material and were processed as a single horizon. For all organic samples, a ~3 cm x 3 cm intact piece of each horizon was removed so as to include the entire vertical length of the horizon. These subsamples were then placed in a 32 oz mason jar with glass beads and perforated aluminum foil lining the bottom of the jar. For each horizon, the subsamples originating from the 3 field sampling locations within a given plot were put into a single jar, creating 1 composite sample per plot, per horizon. For each 10 cm mineral soil core, a small portion of soil was removed using a spatula, incorporating soil from the entire vertical length of the core. Once in jars, deionized water was added to each sample to approximate field capacity and samples were placed in the dark at 15°C. Soil carbon dioxide (CO₂) respiration was measured using an automated CO₂ flux system (Bracho et al., in prep.) for 90 days. The remaining portion of each core was re-frozen immediately after removal of the incubation subsample. These samples were later thawed and a second subsample was removed and placed in a jar using the same method described above. These samples were incubated in the dark at 15°C for 30 days to estimate net N mineralization and nitrification. Within the last 20 days of the experiment, a subset of each sample type was analyzed for ¹⁴CO₂ to determine the age of the C being respired by the different species and soil horizons (Schuur et al. 2009).

The remaining material for each organic horizon and mineral core were homogenized and a composite subsample was made for each plot. This composite sample was then analyzed for moisture content, percent C and N, and pH using the same methods described for Task 3 (above). Exchangeable base cations were measured as outlined in Robertson et al. (1999), using 50 mL of 1 M ammonium chloride (NH₄Cl) mixed with 5 g of field moist organic horizon sample and 10 g of mineral soil, respectively. After shaking for 1 hr on a shaker table, samples were filtered through a GF/A filter via vacuum filtration and frozen until analysis at the Louisiana State University AgCenter (Baton Rouge, LA) on a Spectro CIROS inductively coupled plasma spectrometer (Germany). A 5 g subsample of the composite soil was air-dried and used for analysis of Mehlich P following Kuo et al. 1996. Initial pool sizes of ammonium (NH₄) and nitrate (NO₃) were measured as outlined in Robertson et al. (1999), using the same method used for exchangeable cations, but using 2 M potassium chloride (KCl) instead of NH₄Cl. Samples were analyzed on an Astoria-Pacific International Autoanalzyer (Clackamas, OR) at the University of Florida. At the completion of both 30- and 90-day incubations the entire incubated sample was homogenized and analyzed for NH₄ and NO₃ as described.

In the laboratory, anion and cation resin bags retrieved in June 2013 were rinsed for 20 seconds with running nano-pure water to remove soil particles. Each bag was placed in a tube with 30 mL of a mixture of 0.1 M hydrochloric acid and 2.0 M sodium chloride and shaken for 1 hour on a shaker table. Extractant was then drip-filtered through a Whatman GF/A filter and frozen until further analysis. This extraction process was repeated two additional times for each resin bag. Ammonium and nitrate concentrations were measured colorimetrically using a segmented flow autoanalyzer (Astoria-Pacific, Inc., Clackamas, Oregon, USA). Phosphate was measured colorimetrically within 1 week of extraction using a spectrophotometer microplate reader (PowerWave XS Microplate Reader, Bio-Tek Instruments, Inc., Winooski, VT) using the ascorbic acid molybdenum-blue method (Murphy and Riley 1962). Resins retrieved in August 2013 were extracted using the same method, but with a single extraction and additional N concentration estimated using relationships developed from the June dataset.
Mesh bags were constructed and later filled with leaf material for the mass-loss foliar litter decomposition experiment. Senesced foliar litter for both birch and spruce was collected in the Murphy Dome area in fall 2011 and additional spruce litter was collected in spring 2012. Brown moss was collected in early summer 2012. Foliar litter was thoroughly homogenized, then dried at 30°C in the laboratory. Approximately 0.8 g of birch litter was added to 12 cm x 12 cm gray fiberglass window screen mesh bags. For black spruce, 1 g of needles were added to 8 cm x 8 cm mesh bags constructed with no-see-um mesh. Additional samples of birch litter were added to no-see-um mesh to act as a mesh control. Brown moss was placed field moist for 1 minute in a conventional microwave to inhibit further growth. Moss was then thoroughly homogenized and dried at 30°C. Subsamples weighting 0.5 g were then inserted into 8 cm x 8 cm no-see-um mesh bags. Birch craft sticks (Loew Cornell Woodsies brand) were purchased and a small hole was drilled in one end of the stick. Sticks were then dried at 30°C and weighed.

Statistical analyses

To identify significant differences in species characteristics and nutrients, we used a nested model design where species was the main effect and block was nested within species (JMP Pro 9.0, SAS Institute, Cary NC). Datasets were transformed when needed to meet normality and homoscedasticity assumptions. N mineralization and nitrification data were not able to be normalized with transformations and therefore a Wilcoxon non-parametric test was used. To analyze moss growth, biomass production, and mortality according to litter manipulation treatments (Task 4.5), we will perform mixed-model analyses of variance (ANOVAs) including blocking (3 areas), and 2 levels of nesting (stand type and plot) in R (R Development Core Team 2012). At the end of the experiment, we will include a repeated measure factor in the model to take in account the 3 years of measurement (2013, 2014 and 2015). We will assess the similarity of initial species composition according to treatment types using a mixed-model multivariate analysis of variance (MANOVA) using the same design as the univariate analyses. We will use ordination methods to analyze temporal changes in moss species composition in the transplants (Legendre and Legendre 1998).

Task 5: Monitor mid-successional wildfire network

To improve our understanding of the impacts of tree species composition shifts on C cycling and storage in Alaska’s boreal forest, we identified mid-successional sites across the interior that fell along a deciduous to conifer gradient. These sites were used to develop allometric equations to estimate biomass of all tree and shrub species common to the interior. Methods are detailed in Alexander et al. (2012).

We have revisited these sites to include more intensive investigation of the linkages between moss, plant litter, and stand structure that will complement our litter manipulation experiment (Task 4). We focused on the understory species composition in 51 mid-successional stands (20-62 years since fire) in 17 burn scars in interior Alaska that were sampled by Alexander et al.
(2012). As of October 2013, 9 new early succession stands in 2 fire scars have been sampled to include a wider range of ages in our dataset.

Understory species and ground cover were obtained using a 50 cm x 50 cm grid and point-intercept sampling with five replicates per stand. A pin was inserted in the ground at each of the 25 sampling points. All vascular and non-vascular plant species and other ground cover types (e.g. leaf litter, bare ground, coarse woody debris) that touched the pin were recorded. We also collected stand structure information (DBH, tree density, deciduous importance value, forest type) and some environmental variables (slope, exposition, pH, OL depth) that were published in (Alexander et al. 2012) or collected in 2014-2015.

Univariate and multivariate statistical analyses will be used to assess 1) how moss abundance and species composition vary among mid-successional deciduous, mixed, and coniferous stands, and 2) evaluate how moss abundance, diversity and species composition can be explained using the environmental variables. To address these questions, we will use mixed-models including the fire scar as a random effect. In the first case, we will compare moss and lichen abundance, species composition and diversity among the five forest types defined in (Alexander et al. 2012) using analyses of variance, indicator species analyses, and unconstrained ordinations. In the second case, we will use linear regressions, multivariate regression trees, constrained ordinations, and variation partitioning to assess the relationship between moss and environmental variables. Analyses will be conducted during 2014 and additional sampling is planned for summer 2014.

Task 6: Tree ring sampling and analysis

Field and Laboratory Methods

In the summer of 2012, we established 36 sites within interior Alaska (Task 6.1). Sites were located in four regions along sections of the 1) Steese, 2) Dalton, and 3) Taylor Highways, north and east of Fairbanks (Figure 6.1). Sites were established in pure black spruce stands located within large, recent fire scars dating from 2004 or 2005 in each region, to enable comparisons between pre-fire tree growth and post-fire regeneration. The selection of these sites was stratified based on broad scale topographic positions affecting drainage (moisture) and incident solar radiation (temperature) (Viereck et al. 1983). Within each region, three sites were selected to represent each category of 1) wet and flat, 2) north-facing midslope (cool), 3) south-facing midslope (warm), and 4) dry and flat positions. Additionally, we established 10 sites at the Poker Flats Research Range (PFRR) located along the Steese Highway in interior Alaska, for planned dendroisotopic analysis (Task 6.2). Sites selection was stratified on aspect, with five sites each on a north facing and a south facing slope. Sites in the Steese and Taylor regions have coarse soils and are relatively well drained, whereas a thick cap of loess characterizes soils along the Dalton. The majority (75-80%) of the boreal forest throughout these regions is underlain by permafrost (Osterkamp and Romanovsky 1999), with the exception of south-facing slopes and floodplains near major rivers (Viereck et al. 1983).

At each site, we characterized topographic and soil characteristics, and measured the structure and composition of the pre-fire stand. We measured latitude, longitude, and elevation with a global positioning system (GPS) receiver, and slope and aspect with an inclinometer and
compass. These variables were used for subsequent calculation of insolation, an index of solar radiation received at a site on the summer solstice (as per Bennie et al. 2006). Organic layer thickness was measured at 10 random points per site. In the field, soil pH was measured for samples of near surface mineral soil (A horizon) from three random points per site, using an electronic pH meter (Hanna Instruments) on samples diluted 1:1 with deionized water. Moisture classes, based on topography-controlled drainage and adjusted for soil texture, were assigned to each site on a six-point scale, ranging from xeric to subhygic (Johnstone et al. 2008). Within each site, we measured the DBH of all the pre-fire trees originally rooted within two parallel 2 m x 30 m belt transects. From these data, we calculated the basal area (m$^2$ ha$^{-1}$ of trees >1.4 m in height) of all the pre-fire black spruce trees. At each site, we collected stem disks at the standard height of 1.4 m from 10 randomly selected pre-fire black spruce trees along the two belt transects. In the laboratory, the disks were sanded with increasingly finer sandpaper (up to 400 grit) to produce clearly visible rings (Task 6.3). Annual rings widths were measured (resolution 0.001 mm) on two radii per stem disk using WinDENDRO software version 2012c (Regent Instruments, 2012).

Environmental conditions are known to influence the relative amount of $^{13}$C to $^{12}$C in individual tree rings. Specifically, stomatal closure due to drought stress increases the relative proportion of $^{13}$C to $^{12}$C. To determine the extent of annual variation in $\delta^{13}$C we randomly selected six trees, three from each the south and north-facing slopes in the PFRR for stable isotope analysis on annual rings. Ring samples were manually separated from the outer thirty years (1974-2003) on two perpendicular radii (2.5 x 2.5 cm) per tree using a dissecting scope, razor, and chisel. Wood from the two radii were homogenized into one sample using scissors to pass through ~0.84 mm mesh. This method ensured minimal sample loss. The samples were then oven-dried at 60°C for 48 hrs. We encased approximately 2.5 mg of each sample in a tin cup for isotopic analysis. We chose to use wholewood for this analysis based on previous results obtained from analyzing $\delta^{13}$C of various wood components. All samples were measured for $\delta^{13}$C using a continuous flow, stable-isotope mass spectrometer at the Light Stable Isotope Mass Spec Lab, Department of Geological Science, University of Florida (Task 6.3). The $\delta^{13}$C values are expressed relative to the Vienna PeeDee belemnite (VPDB) standard (Coplen 1995).
Statistical analysis

To prepare the ring width measurements for growth-climate analyses, we visually cross-dated each tree-ring series against master chronologies developed for each site and region. We then used COFECHA version 6.06, which uses statistical measures of similarity or dissimilarity to assess the quality of the cross-dating and measurement accuracy (Grissino-Mayer 2001). All remaining data analyses were conducted in R version 3.0.2 (R Development Core Team 2011) (Task 6.4). To remove the non-climatic trends in ring growth, raw ring width measurements were detrended using a smoothing spline, with a frequency response of 0.5 at a wavelength of 0.67*n years (Zang 2010) in the package ‘dplR, version 1.5.7’ (Bunn 2010). A dimensionless ring width index for each series was produced by dividing the actual ring-width measurement by the curve-fitted value in each year (Cook and Briffa 1990, Bunn 2010).

To assess how site types varied in their growth responses to climate, we created mean chronologies based on site type (dry flat, wet flat, north facing slope, and south facing slope) in the package ‘dplR, version 1.5.7’ (Bunn 2010). General chronology statistics of mean correlation between trees (R), mean sensitivity (MS), signal to noise ratio (SNR), and autocorrelation (AR) for each of the standard chronologies developed per site type and region were calculated to ensure that all the chronologies were suitable for growth-climate analyses (Speer 2010). We then determined the growth-climate responses of each site type using the package ‘bootRes, version 1.2.3’ (Zang 2010), by calculating bootstrapped correlations between

Figure 6.1. Location of thirty-six study sites (black dots) within three regions of recently burned forest (grey), along the Dalton, Steese, and Taylor Highways in interior Alaska.
ring widths and mean monthly temperatures and total monthly precipitation for the period 1975 to 2003. Climate data were obtained from SNAP (Scenarios Network for Alaska and Arctic Planning 2013), which uses Global Climate Model (GCMs) and a PRISM (Parameter-elevation Regressions on Independent Slopes Model) approach (Daly et al. 1997) to downscale climate data to a 1 km resolution. We used averaged downscaled climate data corresponding to each of the four burn scars examined in this study. Growth-climate correlations were calculated using a 17-month climate window, extending from April of the year preceding growth to August of the current year of growth (Fritts, 1976), over the period 1975 to 2003. The significance of each of the 34 climate correlations was tested using the 95 percentile range (Zang 2010).

To confirm our growth-climate responses using stable isotope analysis, we created mean $\delta^{13}C$ chronologies based on site type (north facing slope and south facing slope) in the package ‘dplR, version 1.5.7’ (Bunn 2010). To assess how climate influenced these chronologies we correlated $\delta^{13}C$ discrimination over the 30-year period with mean monthly temperature using the same methods as for growth-climate correlations (see above).

**Task 7: Monitor invasive species in wildfire and management plots**

Fires may create windows of opportunity for the colonization and spread of non-native species. However, fire conditions are heterogeneous and this heterogeneity plays an important role in driving post-fire community assembly (Turner et al. 1999). We conducted observational and empirical studies aimed at determining the effects of fire on current distributions and potential recruitment of non-native plants in Alaskan black spruce forests. Specifically, we surveyed an equal number of burned ($n = 33$) and mature ($n = 33$) black spruce stands along major roadways in interior Alaska to test the effect of fire on non-native plant occurrence and assess whether there are stand characteristics that predict the observed distribution of non-native plants. We also conducted seeding experiments with three non-native species in burned and mature forest to test how variations in substrate types affect seedling germination.

**Non-native plant surveys**

Surveys of non-native plants focused on forests dominated by black spruce and located along two major roadways in interior Alaska. Initial surveys were conducted in 2011 (Task 7.1) and detailed surveys were performed in July 2012 (Task 7.2). Sites were divided equally between recently burned and mature (estimated > 60 years old) forest stands. Roadside samples were selected within areas that had frequent occurrences of non-native plant species along the road verge to provide a potential seed source for colonization. Fifty sites (25 burned, 25 mature) were surveyed adjacent to the Dalton highway north of the Yukon River (burn year 2004), and 16 (8 burned, 8 mature) sites were surveyed adjacent to the Parks highway south of Nenana (burn year 2006). Sampling at each site was based on a 100 m x 2 m belt transect running into the forest from the road verge edge, perpendicular to the road. We recorded the presence of non-native plants along the road verge, the density of individual non-native plants found within the belt transect, and the presence of any non-native plants encountered at the site but outside the sample transect. We also recorded seedling densities of black spruce, Alaska paper birch, and trembling aspen (*Populus tremuloides*) in five randomly located, 1 m x 1 m plots along the transect. Soil measurements of OL depth, thaw depth, and mineral soil pH were made at three randomly selected points along the transect.
Experimental seedling trials

Seeding trials with non-native plants were conducted in the Caribou-Poker Creeks Research Watershed (CPCRW), located approximately 50 kilometers north of Fairbanks, Alaska along the Steese highway (Figure 6.1). Experimental plots were located in a recently burned (2004) and mature black spruce forest. The seeding trials used seed from three non-native plants common in interior Alaska that differed greatly in average seed mass (estimated in the lab from the weight of 100 seeds): *Vicia cracca* L. (bird vetch, 0.012 g·seed⁻¹), *Melilotus officinalis* (0.0018 g·seed⁻¹), and *Taraxacum officinale* G.H. Weber ex Wiggers (common dandelion, 0.00081 g·seed⁻¹). Seeds for the experiment were collected in September 2011 from established populations in Fairbanks, Alaska. Seeding plots were laid out in a stratified random design in adjacent areas of mature and recently burned black spruce forest. Plots were 15 cm x 15 cm in size and clustered in blocks at 10 m intervals along a 100 m transect in each forest type (n = 10 per treatment). Seeds were sown in plots representing five substrate types characteristic of recently burned forests: broadleaf litter, grass litter, regenerating moss (largely *Ceratodon purpureus* and *Polytrichum juniperinum*), charred OL, and exposed mineral soil. There was no exposed mineral soil in the burned forest at the time of the experiment, so mineral soil plots were created by hand removal of surface organic soil and plant litter. Each seeded plot was covered with a mesh cage of galvanized metal (mesh size = 0.5 cm) to prevent small animals from removing seeds or seedlings. Ten seeds of each species were sprinkled on the surface of each plot on 4 June 2012. Total germination counts were made on 30 July 2012, and then all seedlings and visible seeds were removed, and the substrate was dug out and removed from the site to prevent any non-native propagules remaining at the site.

Managed areas

Twelve sites were sampled for invasive plant presence in shearbladed (n = 11) and thinned (n = 1) forests and adjacent unmanaged forest within 150 km of Fairbanks, Alaska in Summer 2012. This sampling was done concurrent with Task 3. Within each site, two 20 m X 2 m transects were placed parallel to each other within the treatments (or mature forests) in a position meant to capture the dominant features of the disturbance. These transects were surveyed for invasive plant presence, and native plant community was measured by recording the presence of all genera that occurred along the transects. The presence of invasive plants that were not along transects was noted, but density estimates were not made. The two transects were also used to visually estimate dominant ground cover types (for a description of ground cover types see sampling protocol for invasive plant surveys above) at ten points on each transect (n = 20).

Statistical analyses

We used information on the presence and absence of non-native plants in burned and mature stands to evaluate the effects of fire on non-native plant occurrence. As *T. officinale* was present at every site with non-native plants, and *C. tectorum* and *M. officinalis* were only encountered in one site each, these analyses are largely predicting the presence of *T. officinale*. We first used a chi-squared test to determine whether the presence of non-native plants was affected by forest type (mature or burned). We then developed models to predict non-native plant presence within burned stands using an information theoretic approach to model selection and inference (Burnham and Anderson 2002) All models were run as binomial linear regression models in the statistical package R 2.14.1. with non-native plant presence as the response variable. We used
Fisher’s exact test to compare germination counts among substrate types in the seeding experiment.

**Task 8: Provision of data for model development, parameterization, and testing**

In this task we are developing, parameterizing and testing two models that have focused on describing various aspects of linkages among climate, fire, and ecosystem structure and function. Specifically, we have incorporated information about combustion of soil OL and permafrost degradation following fire (see immediately below). We are currently incorporating information from the project on vegetation succession and the re-accumulation of the vegetation and soil organic C following fire (see Conclusions for this task). Our approach for incorporating information about the combustion of the (OL) and effects on permafrost degradation involved the development of a predictive module of post-fire reduction of OL thickness. We used existing observations of the OL thickness in recently burned black spruce forests of interior Alaska (Figure 8.1) along with information about topography, weather and fire history to develop a predictive model of the relative loss of OL (ROL) due to fire. ROL refers to the ratio between the loss of OL thickness and the pre-fire OL thickness:

\[
ROL = \frac{OLT_{pre} - OLT_{post}}{OLT_{pre}}
\]

where \( OLT_{pre} \) and \( OLT_{post} \) are the pre-fire and post-fire OL thickness, respectively. The data used for this analysis are from 178 black spruce dominated stands in interior Alaska. Data were stratified into three landscape categories representing different drainage conditions: flat lowland (poorly drained), flat uplands (well drained), and slopes (very well drained, depending on slope angle and aspect, for the influence of runoff and light regime). These landscape categories were assessed from field observations of local topography. The sampled sites covered 31 fire events that occurred between 1983 and 2005 (Turetsky et al. 2011b). The loss of OL was quantified using adventitious roots as a marker for pre-fire OL thickness (Kasischke et al. 2008), combustion rods (Ottmar and Sandberg 2003), or comparison of OL thickness in paired burned versus unburned stands (Harden et al. 2006). Further details on the database can be found in Turetsky et al. 2011b) and the data used in the present analysis have been archived in the Bonanza Creek Long Term Ecological Research database (Kasischke 2013).
Figure 8.1. Location of the sampled sites and the area of simulation in interior Alaska.
We used the database information to then assess the effects of topography and hydrology on ROL using the National Elevation Dataset of the US Geological Survey (USGS) at 60-m resolution (NED, http://ned.usgs.gov/). The topographic descriptors tested included slope, aspect, log-transformed flow accumulation, compound topographic index, curvature, and relief index. The aspect was split into two gradients: the north/south gradient equal to the sine of the aspect and the east/west gradient equal to the cosine of the aspect. The flow accumulation is derived from a cumulative count of the number of pixels that naturally drain into outlets. Sites with high flow accumulation may represent stream channels and cells with accumulation of 0 are local topographic highs. The compound topographic index, which is a steady state indicator of wetness, is a function of slope and flow accumulation (Gessler et al. 1995). The curvature is the second derivative of the surface geometry. Positive curvature indicates a convex surface or divergence of flow. Negative curvature indicates a concave surface or a convergence of flow. The relief index was the standard deviation of elevation estimated for each grid cell and its eight direct neighbors. The effect of weather characteristics on ROL was represented using estimates of monthly mean temperature, summer precipitation, mean vapor pressure, and mean surface downwelling shortwave radiation from 1901 to 2009 at 1x1 km spatial resolution, all of which were downscaled using the delta method (Fowler et al. 2007) by the Scenarios Network for Alaska and Arctic Planning (SNAP, http://www.snap.uaf.edu/), from 0.5 x 0.5 degree resolution data of the Climatic Research Unit (CRU, http://www.cru.uea.ac.uk/). Additional weather characteristics were estimated from DOS-TEM: the maximum ratio between real and potential evaporation of the year prior to fire (ET and PET respectively), the vapor pressure deficit, and soil temperature and soil moisture in the OL. The ratio between the maximum monthly ET and PET (ET/PET) of the previous year was used as an indicator of water stress that occurred the year prior to fire. Information on fire history from 1950 to 2006 (annual area burned, date of burn) was obtained from the Alaska Interagency Coordination Center Large Fire Scar database (AICC, http://fire.ak.blm.gov/; see Kasischke et al. 2002).

Because a number of observations were collected within the same fire scar and in the same landscape category, the observations of ROL were averaged per landscape category within every fire event to avoid pseudo-replication bias, and this reduced the total number of observations from 178 to 49. Additionally, predictor variables were numerous (8 variables related to topographic conditions, 7 variables related to weather conditions and 2 variables related to fire characteristics) and inter-related. For instance, the Pearson correlation between the date of burn and the area burned were significant and equal to 0.401. Similarly, the correlation between soil temperature and soil moisture was -0.406. Therefore, the predictive model of ROL was estimated using partial least square regression (PLS, Tenenhaus et al. 2005), a statistical technique particularly well suited to analyze a large array of related predictor variables, with a limited sample size (Carrascal et al. 2009). The selection of the predictors to eliminate from the analysis was based on the regression coefficient of the centered and scaled data and on the predictor’s importance. The data were centered and scaled to have a null mean and a standard deviation of 1 to ensure that each variable had the same weight and the selection of the predictors was based on how much variation of the response a predictor explains. The predictor’s importance summarizes its contribution to the whole model (its capacity in fitting the model for both X the predictors and
Y the response), weighing the contribution of each predictor according to the variance explained by each PLS component. If a predictor has a relatively small regression coefficient and a small value of importance (less than 0.8, see Wold 1994), then it is a prime candidate for deletion.

The predictive model of ROL identified the landscape category as the primary driver of the relative amount of OL burned during fire. Therefore, we developed separate continuous predictive models of ROL for each landscape category. All analyses were performed using the SAS statistical package (SAS 9.1, SAS Institute Inc., Cary, NC, USA). The assumptions of normality and homoscedasticity were verified by examining residual plots. Effects were considered highly significant at the 0.05 level and marginally significant at 0.10.

**Task 9: Forecast future landscape distribution with coupled models**

The research in this task involves four components: (1) model development and testing of the terrestrial ecosystem model (TEM) and the model of fire dynamics (the Alaska Frame-based Ecosystem Code, ALFRESCO), (2) coupling of these two models together, (3) evaluating the performance of the coupled models, and (4) projection of the future distribution of vegetation and permafrost. To date, we have made substantial progress on model development and testing of TEM and ALFRESCO (see the Results and Discussion section below). We are currently conducting a regional simulation in which we are asynchronously coupling the models together (i.e., without feedback between the models). The outputs from this simulation will serve as a reference to evaluate the effects of feedbacks between the models from the synchronous coupling. See the Conclusions section below for further details on our schedule for the synchronous coupling, evaluation of feedback effects, and the projection of future distribution of vegetation and permafrost.

**Approach to TEM Development and Testing**

The predictive model of ROL was integrated into the disturbance module of the Dynamic Organic Soil version of the Terrestrial Ecosystem Model (DOS-TEM). The TEM family of process-based ecosystem models has been designed to simulate C and N pools of the vegetation and the soil, and C and N fluxes among vegetation, soil, and the atmosphere (Raich et al. 1991; McGuire et al. 1992). Previous regional applications of TEM in northern high latitudes have investigated how biogeochemical dynamics of terrestrial ecosystems in these regions are affected at seasonal to century scales by processes like soil thermal dynamics (Zhuang et al. 2001; Zhuang et al. 2002; Zhuang et al. 2003), snow cover (Euskirchen et al. 2006; Euskirchen et al. 2007), and fire (Balshi et al. 2007; Yuan et al. 2012). DOS-TEM has been primarily developed to represent how changes in organic horizon thickness affect ecosystem processes immediately following fire and during stand succession. During stand succession, soil organic thickness is computed after soil C pools are altered by ecological processes (i.e. litterfall, decomposition) based on the relationships between soil C content and soil organic thickness of different organic horizons in black spruce stands in Manitoba, Canada (Manies et al. 2005; Manies et al. 2006; Yi et al. 2009a,b):

\[ C = a * x^b \]  
(2)
where $C$ and $x$ are $C$ content ($g \text{ C cm}^{-2}$) and thickness (cm) of the OL, and $a$ and $b$ are fitted parameters (see Yi et al. 2009a for details). The vertical distribution of soil $C$ in the mineral layers is determined from a combined exponential relationship between depth and $C$ density:

$$C_{\text{dens}} = a + b \times \text{depth} \times \exp^{c \times \text{depth}}$$

where $C_{\text{dens}}$ is the carbon density ($kg \text{ m}^{-3}$), depth is the soil depth (cm) from the boundary between the mineral and organic layer and $a$, $b$ and $c$ are fixed parameters (10.056, 6.147, -0.0662 respectively). This relationship has been parameterized using observations from study sites in interior Alaska (Harden et al. 2012a; Johnson et al. 2011). The predictive model of ROL has been added into the disturbance module of DOS-TEM, which calculates the proportion of organic layer thickness and vegetation biomass that is burned, the proportion of vegetation biomass that died from the fire, and the $C$ and $N$ lost in combustion emissions. This module is called each year a fire occurs. In the previous version of DOS-TEM, the ROL was estimated using a look-up table for two classes of drainage (lowlands and uplands, Yi et al. 2010). We replaced this look-up table by the continuous algorithms developed from the PLS analysis. The post-fire OL thickness is then calculated from the pre-fire thickness and the estimated ROL. The post-fire $C$ content in the OL is then updated using a power relationship between OL thickness and $C$ content from equation (2) (see Yi et al. 2009 for details). The calibration of DOS-TEM makes use of (1) soil $C$ observations from study sites in interior Alaska (Harden et al. 2012a; see Yi et al. 2010) and from the National Soil Carbon Database for interior Alaska (Johnson et al. 2011, see Yuan et al. 2012), and (2) vegetation $C$ and $N$ pools and $C$ fluxes from studies conducted by the Bonanza Creek Long Term Ecological Research (LTER) program (Ruess et al. 1996 for lowland black spruce forest and Vogel et al. 2005 for upland black spruce forest; see Yi et al. 2010 and Yuan et al. 2012). In this study, the integration of the predictive model of fire severity into DOS-TEM was verified by running DOS-TEM for each of the ROL observation sites used to develop the predictive models of fire severity.

We used DOS-TEM with the integrated predictive model of fire severity to assess the relative importance of climate warming, fire regime and local topography on organic layer, active layer thickness, and soil $C$ dynamics across the landscape in interior Alaska. The area of simulation overlaps all the fire events where field observations of organic layer reduction had been collected. This study area covers about 240,000 km$^2$ in the eastern part of the Yukon River Basin within Alaska (Figure 8.1). The distribution of landform categories over the area of simulation was assessed from topographic information (see Supplemental material in Genet et al. 2013). Based on DOS-TEM outputs, active layer dynamics were analyzed by quantifying the changes in the active layer thickness in relation to changes in the OL thickness over time. Changes in soil $C$ stocks from DOS-TEM were evaluated by quantifying changes in total soil $C$ stocks over time. To evaluate the effects of climate warming and changing fire regime, we conducted a factorial analysis combining a scenario with and without warming (Figure 9.1a) and a scenario with changing or constant fire regime (Figure 9.1b).
Figure 9.1. Time series of a) the CCCMA warming and b) the CCCMA-ALFRESCO fire scenarios: the red lines represent no warming and no fire intensification scenarios and the black represent warming and fire intensification scenarios.

The warming scenario combined (1) historical climate variability from 1901 to 2009 with (2) climate variability from 2010 to 2100 projected by the Coupled General Circulation Model version 3.1 - t47 developed by the Canadian Centre for Climate Modeling and Analysis (CCMA, http://www.cccma.ec.gc.ca/data/cgcm3/) for the mid-range emission scenario A1B. The climate data were bias-corrected, downscaled and re-sampled by the SNAP using the same methods as for the historical climate data for site-specific simulations (see Task 8). For the no-warming scenario, a smoothing cubic-spline with an amplitude of frequency response of 0.5 and a periodicity of 50 years was fitted to the time series of mean air temperatures. The detrended air temperatures were then computed for each 1-km grid cell as a function of the mean air temperature from 1901 to 1930 \( (T_{[1901-1930]}) \), and the difference between the current \( (T_{curr}) \) and predicted air temperature \( (T_{pred}) \) from the smoothing spline:

\[
T_{air} = T_{[1901-1930]} + (T_{curr} - T_{pred})
\]

The detrending procedure was applied to the time series of air temperature for each 1-km grid cell for every month of the year. The scenario with changing fire regime combined (1) historical records from 1950 to 2009 and (2) projected scenarios from the ALaska FRame-based EcoSystem Code (ALFRESCO1.0, Rupp et al. 2001; Rupp et al. 2002). This scenario represents the changes in fire frequency in response to climate change and changes in vegetation composition over time. The normalized fire regime data set was generated by using a fire return interval (FRI) data set that was developed from the 1960-1989 fire data (Yuan et al. 2012). This scenario represents a constant fire frequency over time that reflect conditions prior to the significant increase of annual area burned observed in Alaska after the 1990’s. For each 1-km grid cell, the first fire year of the normalized fire regime dataset was randomly chosen between 1900 and 1900+FRI. The next fire years then occurred at regular FRI.

The effects of warming, fire regime and local topography and their interactions with the dynamics of active layer thickness and soil C balance were statistically analyzed for the sampled
sites (Turetsky et al. 2011), using an analysis of variance. The absence of spatial autocorrelations for the observed ROL was checked prior to the analysis based on two indices (Moran’s $I = -7.53 \times 10^{-4}$ and Geary’s $c = 1.009$, $p = 0.624$). The effects of warming, fire and topography were tested by examining the changes in organic layer thickness, active layer thickness and soil C stocks, between the early 20th century and the late 21st century. We computed the means of these variables for the beginning of the 20th century [1901-1930] and the end of the 21st century [2071-2100], and then took the difference between these two periods to compare pre-industrial conditions with changes that occurred during the entire period studied.

Approach to ALFRESCO Development and Testing

Model development activities focused on ingestion of driving climate datasets and partitioning regional fire sensitivity parameters. CMIP3 monthly air temperature and total precipitation data was downscaled to 1km spatial resolution for historical and the five best performing general circulation model (GCM) projections in Alaska. Climate scenarios for the three primary emission scenarios (B1, A1B, and A2) for each model were downscaled. Additionally, surface downwelling shortwave radiation and vapor pressure fields were also downscaled (historical and projected) for ingestion into the TEM simulations (see Task 8). CMIP5 data for all fields has been prepared and downscaled and will be available for future project use and model simulation.

Regional fire sensitivity parameters were developed to modify fire ignition potential at the ecoregion level. This utility provides capacity to represent and calibrate broad scale differences in climate-ecosystem interactions across the entire state of Alaska. Additionally, this utility allows for any functional and spatially explicit partitioning of fire routine parameters. This utility will be utilized to complete the fire management scenarios identified in task 10 – specifically simulating model response to changing fire management zones within DoD training lands.

Preliminary model testing and calibration has been completed using a statewide simulation domain for the historical period (1900-2010) and projections (2011-2099). Each climate scenario (five GCMs each with three emission scenarios) was simulated 200 times ($n = 3000$). Additional calibration and testing is currently focused on the interior portion of the simulation domain containing the majority of DoD training lands in Alaska.

Task 10: Planning and participation in workshops for Alaska land managers

Alaska holds an unprecedented level of public responsibility for its land-based resources -70% of the state is under the jurisdiction of federal and state resource managers. Fire management in Alaska is multifaceted as federal and state land managers seek ways to manage fuels and fire to both improve community safety and meet multiple natural resource objectives. Distribution of information to fire and land managers poses a major challenge to effective fire science delivery. To meet this challenge, we have met with fire and land managers in Interior Alaska three times during the last year to identify what applications of our framework would produce useful information for fire management and land planning. These discussions have led to the identification of three questions that we will address. Our activities in addressing each of these questions will lead to several information products that we will deliver to the managers in 2015.
We summarize the three questions and the strategy for addressing each question the Results and Discussion section below (Task 10).
Results and Discussion

Task 1: Field sampling of wildfire site network

Across the sites measured in 2005 and 2011, the mean organic layer depth was 3 cm less in 2011 and the thaw depth had increased by 17 cm (Figure 1.1). These results suggest that in the 6 years following the severe 2004 fire year, there has been no re-accumulation of soil organic matter and possibly a small loss. In general, there has also been permafrost thaw. We are continuing to explore the relationship between organic layer depth and permafrost across sites using statistical models that will allow us to assess inter-site variability in responses, with a focus on important environmental variables including the proportion of organic material lost during the fire and site drainage class.

Preliminary results from our seedling analyses show a large range of variation in total and new, current year biomass among species individual seedlings (Figure 1.2). This puts us in a very good situation to explore the factors that are likely to explain this variation, including hypothesized effects of fire severity and post-fire organic layer depth on seedling growth.

Figure 1.1. Change in organic layer and thaw depths for 6 years following the 2004 severe fire year.
Figure 1.2. Boxplots representing preliminary total biomass (left panel) and new growth biomass (right panel) of each species planted in the intensive sites. BS = black spruce, WS = white spruce, TA = trembling aspen, LP = lodgepole pine, PB = paper birch. Each colour represents a different year of field measurement. Boxes encompass the 25%–75% quartiles of the data, with the median indicated by the thick line through the center of each box. Whiskers extending from the box encompass the 95% quartiles, and extreme observations are shown as black open circles. Values for unharvested trees were obtained using allometric equations created from 2011 harvested trees. Note that the axes are different for each panel.
Task 2: Identification and sampling of new wildfire plots and unburned control plots

For the hardwood stands included in the burned/unburned pair study, the organic layer was reduced by a mean of 7 cm during fire, resulting in a residual organic layer that was 1-3 cm thick. Aboveground live tree biomass losses with fire varied between the hardwood stands, with one site showing complete tree mortality and the second site losing 30% of live tree biomass relative to adjacent unburned forest. Both burned areas showed very high combustion rates, with < 5% of fine branch material remaining on all measured trees.

Across black spruce stands of varying topographic positions, we found that fires rarely consumed C that had accumulated longer than 10-20 years ago (Figure 2.1). At the organic layer-mineral soil interface, the age of C ranged from 150 to > 600 years old, suggesting that in the boreal forest at least a portion of the organic layer may be retained through numerous fire cycles.

![Figure 2.1. Estimated age of the organic layer surface and deepest organic soils for 9 cores sampled across 4 recent fires in interior Alaska.](image-url)
Task 3: Identification and sampling of fire management plots

Aboveground live tree biomass C was significantly smaller in managed areas relative to adjacent unmanaged black spruce forest (Figure 3.1). Results suggest that thinning reduced tree C stocks by a mean of 2 kg C m$^2$ and shearblading reduced C stocks by 2.6 kg C m$^2$. The mean organic layer depth was 24 cm in unmanaged forest, 20 cm in thinned stands, and 10 cm in shearbladed areas. Soil organic layer C stocks were significantly smaller in the shearbladed area relative to the thinned and control soils (Figure 3.2), however the C stocks were higher than would be expected given the large difference in organic layer thickness. This was explained by a greater bulk density in shearbladed organic layers, which indicates potential compaction of the soil by machinery used during management. Collectively, soil and aboveground tree biomass losses due to management were 2 kg C m$^2$ and 4.5 kg C m$^2$ from thinned and shearbladed areas, respectively, when compared to unmanaged forest. These findings indicate the forest management can have profound effects on ecosystem C stocks. There was a trend for increased thaw depth in thinned areas and a significant increase in thaw depth in shearbladed stands relative to unmanaged forests (Figure 3.3). Thinning increased thaw depth by an average of 16 cm and shearblading by 42 cm. We had hypothesized that loss of the organic layer could induce permafrost degradation and our results suggest this may be occurring. These findings indicate that forest management, especially shearblading, many reduce ground stability, which could have implications for infrastructure and military training lands.

Tree seedling density was highest in shearbladed areas, with an observed mean of 7 seedlings m$^2$ (Figure 3.4). Deciduous tree species seedlings were most common, however some black spruce seedlings were also present. Thinned and unmanaged forests averaged < 1 seedling m$^2$. This result suggests that shearblading may facilitate deciduous forest establishment in areas previously dominated by black spruce.

![Figure 3.1. Live tree aboveground biomass in unmanaged forest (control), thinned, and shearbladed areas. Presented data is mean (SE) for all plots within each treatment.](image-url)
Figure 3.2. Mean (SE) organic layer carbon stocks for control and managed study areas.

Figure 3.3. Mean (SE) thaw depth for unmanaged (control), thinned, and shearbladed managed areas studied across interior Alaska.
Task 4. Moss and litter manipulation experiment

Aboveground

The mean age of trees across the study site was 45 years, indicating forest establishment following the 1958 fire. Within birch plots, 84% of the live trees were birch and 99% of the total aboveground biomass was the target plot species. Black spruce trees comprised 91% of all stems in the spruce plots and accounted for 83% of the total aboveground biomass. Total live tree biomass and basal area were significantly higher in birch stands relative to spruce (Table 4.1).

Fresh birch foliage exhibited significantly higher concentrations of N, P, Ca, Mg, and K than current year black spruce needles (Table 4.2). Black spruce showed a significantly larger C:N ratio and higher C concentration than birch. A comparison of current and previous years’ growth of black spruce needles indicated significantly higher P and K concentrations in current year’s growth and significantly higher Ca in previous years’ growth (Table 4.2). Foliar litter showed a trend for higher N, P, Mg, and K concentrations in birch while current year spruce needles exhibited a larger C:N ratio and higher Ca concentration (Table 4.2). Total litter inputs were significantly higher in birch plots relative to spruce (Table 4.2). In birch plots, 76% of the incoming litter was birch foliar litter, while 50% of litter inputs in spruce plots were spruce needles. Branches and bark contributed to the total inputs in both stands, accounting for 21% and 17% of total litter inputs in birch and spruce, respectively. Seeds, cones, and miscellaneous components also made minor contributions.

Moss transplant and deciduous litter manipulation

The initial survey of the 2013 moss growth data did not show any clear patterns for the 2012 shoots (Figure 4.3) or for the 2013 shoots (data not shown). There was no clear effect of either
forest type or treatment on the width of the 2012 shoots, although there was a trend toward longer shoots in 2012 in the litter exclusion treatments (Figure 4.3). The growth in width and length of the 2013 shoots was higher in birch stands than in spruce stands, without a clear treatment effect of the litter manipulation. Initial moss species composition among the different treatments was very similar. Overall, *H. splendens* covered about 85% of the transplants and *Pleurozium schreberi* was the second most common species, covering about 38% of the transplants. Note that these species can occur together at the same grid point, and thus total cover is higher than 100%.

**Soils**

Organic layer concentrations of N, Mehlich extractable P, exchangeable Ca, and exchangeable Mg were significantly higher in birch soils (Table 4.3). The C:N ratio was significantly larger in spruce soils and there was a trend for higher C concentration in the black spruce organic layer, but this was not statistically significant. Soil pH was significantly higher in birch soils while exchangeable K and bulk density did not differ between the two stand types. The mass of the soil organic layer was significantly larger in black spruce relative to birch (9.7 ± 0.3 vs. 5.2 ± 0.9, P < 0.0001), which contributed to significantly larger stocks of both C and N in the organic layer of black spruce plots (Figure 4.4). Mehlich extractable P stocks were also significantly higher in spruce plots (1.08 ± 0.14 vs. 0.86 ± 0.15, P = 0.03). Exchangeable K stocks were significantly larger in birch soils and no differences in Ca or Mg stocks were observed (data not shown). These results indicate that despite higher concentrations of N, Mehlich P, Ca, and Mg in the birch organic layer, these nutrients is not accumulating in the organic layer, suggesting more rapid turnover in birch soils.

In the uppermost 10 cm of mineral soil, black spruce exhibited a significantly higher C:N ratio as well as higher concentrations of both C and N (Table 4.4). Exchangeable Mg concentrations and pH were also significantly higher in black spruce. In contrast, birch mineral soils contained significantly higher concentrations of Mehlich extractable P and exchangeable K (Table 4.4). No differences were observed in bulk density or exchangeable Ca, although there was a trend for higher Ca in black spruce soils. Concentrations of total P, Ca, Mg, and K were similar between the two forest types, although a trend for higher total P was observed in black spruce (Table 4.4). Mineral soil stocks of C and N in the 0-10 cm depth increment were significantly larger in black spruce while stocks of Mehlich P and exchangeable K were higher in birch. No significant differences were observed in stocks of exchangeable Ca and Mg.

Net N mineralization was significantly higher in birch organic layer soils after both 30 and 90 days (Figure 4.5). Within the mineral soils, significantly higher net N mineralization was only observed after 90 days, but there was a trend for higher rates in birch after 30 days. Similarly, significantly higher rates of net nitrification were observed in the organic layer of birch soils after 30 and 90-day incubations (Figure 4.5). No differences in net nitrification were observed in the mineral soils for either time period.

Ion resins installed in the organic layers of birch soils showed significantly higher accumulation of both NH$_4^+$ and NO$_3^-$ than in spruce soils (Table 4.5). Mineral soil NH$_4^+$ accumulation was significantly higher in birch and no differences were observed between the species for NO$_3^-$. Resin sorbed P concentrations were significantly higher for birch in both organic and mineral soils (Table 4.5).
Table 4.1. Live tree aboveground biomass and basal area for studied plots at Murphy Dome.

<table>
<thead>
<tr>
<th>Live Tree Biomass</th>
<th>Alaska Paper Birch</th>
<th>Black Spruce</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Plot species kg m⁻²</td>
<td>11.0(1.38)</td>
<td>4.13(0.86)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>Other species kg m⁻²</td>
<td>0.12(0.10)</td>
<td>0.88(0.07)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>Total kg m⁻²</td>
<td>11.12(1.33)</td>
<td>5.01(0.80)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>Total basal area cm² m⁻²</td>
<td>28.62(2.65)</td>
<td>21.73(2.52)</td>
<td>0.0043*</td>
</tr>
</tbody>
</table>

Mean(SD) n = 3 blocks where * indicates significant difference between forest types.
Table 4.2. Foliar chemistry and litter chemistry and inputs for each stand type.

<table>
<thead>
<tr>
<th></th>
<th>Alaska Paper Birch</th>
<th>Black Spruce</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Current year growth</td>
<td>Old growth</td>
<td></td>
</tr>
<tr>
<td><strong>C:N</strong></td>
<td>18.75(.56)</td>
<td>46.61(3.97)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td><strong>% C</strong></td>
<td>48.00 (0.63)</td>
<td>50.01 (0.12)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td><strong>% N</strong></td>
<td>2.57 (0.11)</td>
<td>1.08 (0.09)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td><strong>% P</strong></td>
<td>0.24(0.004)</td>
<td>0.11(0.01)a</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td><strong>% Ca</strong></td>
<td>0.70(0.08)</td>
<td>0.30(0.03)a</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td><strong>% Mg</strong></td>
<td>0.37 (0.03)</td>
<td>0.09(0.01)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td><strong>% K</strong></td>
<td>0.86 (0.03)</td>
<td>0.49(0.04)a</td>
<td>&lt; 0.0001*</td>
</tr>
</tbody>
</table>

**Foliar Litter**

| **% N**                  | 0.92(0.02)         | 0.51         |
| **% C**                  | 49.39(0.17)        | 49.26        |
| **C:N**                  | 54.46(1.02)        | 97.46        |
| **% P**                  | 0.21(0.01)         | 0.03         |
| **% Ca**                 | 0.96(0.04)         | 1.74         |
| **% Mg**                 | 0.44(0.02)         | 0.04         |
| **% K**                  | 0.41(0.04)         | 0.10         |

**Litter Inputs g m⁻² yr⁻¹**

<table>
<thead>
<tr>
<th></th>
<th>Total OM</th>
<th>All foliage</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Total OM</strong></td>
<td>242.8(20.7)</td>
<td>93.8(19.5)</td>
<td>P &lt; 0.0001*</td>
</tr>
<tr>
<td><strong>All foliage</strong></td>
<td>178.4(8.8)</td>
<td>72.2(15.7)</td>
<td>P &lt; 0.0001*</td>
</tr>
<tr>
<td><strong>Spruce foliage</strong></td>
<td>178.1(8.6)</td>
<td>27.1(8.7)</td>
<td></td>
</tr>
<tr>
<td><strong>Bark and branches</strong></td>
<td>52.2(27.8)</td>
<td>16.2(9.2)</td>
<td></td>
</tr>
<tr>
<td><strong>Seeds</strong></td>
<td>7.6(3.0)</td>
<td>1.0(0.3)</td>
<td></td>
</tr>
<tr>
<td><strong>Pollen cones</strong></td>
<td>2.6(3.3)</td>
<td>2.4(1.9)</td>
<td></td>
</tr>
<tr>
<td><strong>Misc. components</strong></td>
<td>2.05(0.2)</td>
<td>2.0(0.5)</td>
<td></td>
</tr>
</tbody>
</table>

**Foliar Litter Inputs g m⁻² yr⁻¹**

| **N**                    | 1.6(0.1) | 0.1(0.04) |
| **C**                    | 88.0(4.5) | 13.4(4.3) |
| **P**                    | 0.4(0.01) | 0.01(0.002) |
| **Ca**                   | 1.7(0.1) | 0.5(0.2)   |
| **Mg**                   | 0.8(0.1) | 0.01(0.003) |
| **K**                    | 0.72(0.1) | 0.03(0.01) |

Mean (SD), n = 3 plots. For the spruce foliar inputs, the nutrient concentrations are from only 1 sample, so no statistics were included. P values and * indicate statistical outcomes comparing the two tree species for the variable indicated in the first column of each row. Different letters for the concentrations of live spruce foliage indicate significant differences between current and old growth needles (P = 0.05)
Figure 4.3: Average moss growth on the 2012 shoot measured during the summer 2013 and standard error as a function of forest type (spruce – dark grey, birch – light grey) and treatments (control, addition, ambient, exclusion). A) Width growth (cm). B) Length growth (cm).
Table 4.3. Soil organic layer element concentrations, C:N, and bulk density for the two stand types.

<table>
<thead>
<tr>
<th>Organic Layer Property</th>
<th>Alaska Paper Birch</th>
<th>Black Spruce</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>%N</td>
<td>1.75(0.07)</td>
<td>1.33(0.10)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>%C</td>
<td>41.07(1.88)</td>
<td>43.01(0.25)</td>
<td>0.0656</td>
</tr>
<tr>
<td>C:N</td>
<td>23.58(1.15)</td>
<td>32.62(2.15)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>Mehlich P mg g⁻¹</td>
<td>0.17(0.03)</td>
<td>0.11(0.01)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>Exchangeable Ca cmol c kg⁻¹</td>
<td>23.65(1.69)</td>
<td>14.72(3.24)</td>
<td>0.0001*</td>
</tr>
<tr>
<td>Exchangeable K cmol c kg⁻¹</td>
<td>2.87(0.13)</td>
<td>2.92(0.35)</td>
<td>0.74</td>
</tr>
<tr>
<td>Exchangeable Mg cmol c kg⁻¹</td>
<td>12.54(1.27)</td>
<td>5.78(2.42)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>pH</td>
<td>4.98(0.19)</td>
<td>4.33(0.12)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>Bulk density g cm⁻³</td>
<td>0.07(0.0004)</td>
<td>0.06(0.007)</td>
<td>0.42</td>
</tr>
</tbody>
</table>

Mean (SD) of weighted organic layer concentrations, n = 3 blocks. * indicates significant differences between forest types.

Figure 4.4. Organic layer C (A) and N (B) stocks for studied Alaska paper birch and black spruce forest. Data presented is mean(SE) and * indicates a statistically significant difference between the species.
Table 4.4. 0-10 cm mineral soil properties and nutrient concentrations for birch and spruce stands.

<table>
<thead>
<tr>
<th>Soil Property</th>
<th>Alaska Paper Birch</th>
<th>Black Spruce</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>% N</td>
<td>0.20(0.06)</td>
<td>0.31(0.08)</td>
<td>0.0003*</td>
</tr>
<tr>
<td>% C</td>
<td>3.65(0.86)</td>
<td>6.41(1.48)</td>
<td>&lt; 0.0001*</td>
</tr>
<tr>
<td>C:N</td>
<td>18.60(1.02)</td>
<td>20.61(0.67)</td>
<td>0.0174*</td>
</tr>
<tr>
<td>Mehlich P mg g⁻¹</td>
<td>0.03(0.002)</td>
<td>0.006(0.005)</td>
<td>0.0002*</td>
</tr>
<tr>
<td>Exchangeable Ca cmol kg⁻¹</td>
<td>3.79(.064)</td>
<td>6.10(4.34)</td>
<td>0.0694</td>
</tr>
<tr>
<td>Exchangeable Mg cmol kg⁻¹</td>
<td>1.30(0.52)</td>
<td>2.39(2.14)</td>
<td>0.0434*</td>
</tr>
<tr>
<td>Exchangeable K cmol kg⁻¹</td>
<td>0.29(0.05)</td>
<td>0.23(0.03)</td>
<td>0.0339*</td>
</tr>
<tr>
<td>pH</td>
<td>5.0(0.1)</td>
<td>5.2(0.4)</td>
<td>0.0178*</td>
</tr>
<tr>
<td>Bulk density g cm⁻³</td>
<td>0.71(0.08)</td>
<td>0.65(0.04)</td>
<td>0.1364</td>
</tr>
</tbody>
</table>

Total Element Concentrations

<table>
<thead>
<tr>
<th>Total Element Concentrations</th>
<th>P ppm</th>
<th>Black Spruce</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>P ppm</td>
<td>636.67(81.45)</td>
<td>773.33(41.63)</td>
<td></td>
</tr>
<tr>
<td>% Ca</td>
<td>1.26(0.32)</td>
<td>1.22(0.29)</td>
<td></td>
</tr>
<tr>
<td>% Mg</td>
<td>0.94(0.15)</td>
<td>0.86(0.09)</td>
<td></td>
</tr>
<tr>
<td>% K</td>
<td>1.26(0.12)</td>
<td>1.26(0.05)</td>
<td></td>
</tr>
</tbody>
</table>

Mean (SD) n = 3 blocks. P values indicate results of full model with block nested within species. For the total element concentrations, also n = 3 blocks, but only 1 sample per block, no statistical analyses were conducted.
Figure 4.5 Net nitrogen mineralization and nitrification for both organic and 0-10 cm mineral soils from both studied forest types, following a 90- day laboratory incubation. Values indicate mean(SD). * indicates significant differences between species for the given rate and depth.

* P = 0.005

* P <0.001
Table 4.5. One-year cumulative ion resin concentrations from organic and mineral soil layers from birch and spruce study plots.

<table>
<thead>
<tr>
<th>Ion Resin ug g⁻¹</th>
<th>Alaska Paper Birch</th>
<th>Black Spruce</th>
<th>P-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>NH₄⁺</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Organic layer</td>
<td>179.1(99.9)</td>
<td>83.7(55.2)</td>
<td>0.04*</td>
</tr>
<tr>
<td>Mineral</td>
<td>115.8(43.3)</td>
<td>22.4(11.8)</td>
<td>0.0008*</td>
</tr>
<tr>
<td>NO₃⁻</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Organic layer</td>
<td>3.0(3.0)</td>
<td>0.3(0.4)</td>
<td>0.02*</td>
</tr>
<tr>
<td>Mineral</td>
<td>6.6(5.3)</td>
<td>31.2(54.3)</td>
<td>0.90</td>
</tr>
<tr>
<td>P</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Organic layer</td>
<td>802.0(217.1)</td>
<td>143.5(90.3)</td>
<td>0.0001*</td>
</tr>
<tr>
<td>Mineral</td>
<td>1251.2(609.9)</td>
<td>140.4(50.5)</td>
<td>&lt; 0.0001*</td>
</tr>
</tbody>
</table>

Mean (SD), n = 3 blocks. * indicates significant differences between the two species for the given ion and sampling layer.
Table 4.6. Results of an indicator species analysis with 4999 permutations. Bold font indicates species that were significantly (p < 0.05) associated with the specified forest type.

<table>
<thead>
<tr>
<th>Species</th>
<th>Forest type</th>
<th>Indicator value</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Aulacomium palustre</em></td>
<td>Black spruce stands (PM)</td>
<td>0.47</td>
<td>0.008</td>
</tr>
<tr>
<td><em>Sphagnum spp.</em></td>
<td>Black spruce stands (PM)</td>
<td>0.34</td>
<td>0.034</td>
</tr>
<tr>
<td><em>Tomenthyphnum nitens</em></td>
<td>Black spruce stands (PM)</td>
<td>0.27</td>
<td>0.110</td>
</tr>
<tr>
<td><em>Pleurozium schreberi</em></td>
<td>Mixed black spruce and Alaskan paper birch stand (PM+BP)</td>
<td>0.39</td>
<td>0.037</td>
</tr>
<tr>
<td><em>Peltigera aphthosa</em></td>
<td>Mixed black spruce and aspen stand (PM+PT)</td>
<td>0.37</td>
<td>0.029</td>
</tr>
<tr>
<td><em>Cladonia</em> spp.</td>
<td>Mixed black spruce and aspen stand (PM+PT)</td>
<td>0.33</td>
<td>0.108</td>
</tr>
<tr>
<td><em>Thuidium abinetum</em></td>
<td>Mixed black spruce and aspen stand (PM+PT)</td>
<td>0.17</td>
<td>0.212</td>
</tr>
<tr>
<td>Unknown acrocarpous moss</td>
<td>Aspen stand (PT)</td>
<td>0.15</td>
<td>0.177</td>
</tr>
</tbody>
</table>
Preliminary results indicate that total moss cover was significantly different among the five mid-successional forest types ($F_{4,31}=17.11$, $p < 0.0001$), i.e. pure black spruce stand, pure Alaskan paper birch stand, pure aspen stand, mixed black spruce and Alaskan paper birch stand, and mixed black spruce and aspen stand. Black spruce stands had the highest moss cover, and the pure Alaskan paper birch stands had the lowest moss covers (Figure 5.1). Mixed black spruce and aspen stands had an intermediate moss cover, while mixed black spruce and Alaskan birch stands had a moss cover that was not significantly different from that of pure black spruce stands. The total lichen ground cover differed among forest types ($F_{4,31}= 4.80$, $p = 0.004$; Figure 5.1). Pure birch stands and mixed birch stands had a significantly lower lichen cover than the pure aspen, mixed aspen and spruce and pure spruce stands. We also found a significant difference in non-vascular plant diversity among the forest types ($F_{4,31}=4.91$, $p = 0.004$). Non-vascular plant diversity was significantly higher in pure black spruce stands and mixed stands than in pure Alaskan birch stands (Figure 5.1). Aspen stands were showing a non-vascular plant diversity that was intermediate between those two groups. Leaf litter cover differed among forest types ($F_{4,31}=15.04$, $p < 0.0001$). The differences among leaf litter cover in forest types were consistent with the results obtained for the total moss cover and with the hypothesis (Figure 5.1). Birch stands had the highest litter cover, and black spruce stands had the lowest litter cover. As the previous results suggested, there was a strong negative relationship between total moss cover and leaf litter cover ($F_{1,34}= 77.30$, $p < 0.0001$; Figure 5.2). There was also a strong relationship between lichen cover and leaf litter ($F_{1,34}= 52.32$, $p < 0.0001$; Figure 5.2).

We found unexpected results regarding the cover of the dominant moss and lichen species in the five forest types. We expected the feather mosses *Hylocomium splendens* and *Pleurozium schreberi* to be very dominant in spruce stands, and to have a few co-dominant species in mixed or deciduous stands. However, we found that moss and lichens species dominance was shared more or less equally between seven to eight moss species in pure black spruce stands (in decreasing order of cover: *Aulacomium palustre*, *Pleurozium schreberi*, *Polytrichum commune*, *Cladonia* spp., *Hylocomium splendens*, *Sphagnum* spp., and *Tomentypnum nitens*). Both types of mixed stands had high cover of *Polytrichum commune* and *Pleurozium schreberi*. Birch had almost no moss or lichen cover. In aspen stands, two species of lichens and *Polytrichum commune* were the most abundant. Those results were further confirmed by the results of the indicator species analyses (Table 4.6). Neither of the pure deciduous stands had an indicator moss or lichen species, which means that no species shown a high abundance and fidelity to those forest types. Mixed black spruce and aspen stands had the lichen *Peltigera aphthosa* as an indicator species, which suggests dry conditions, while the indicator species of mixed black spruce and Alaskan paper birch was *Pleurozium schreberi*, a very common feather moss. Finally, the two indicator species of pure black spruce stands are mosses that are usually found in wet and acidic environments. We were expecting *Hylocomium splendens* to be selected as an indicator species. The fact that it was not may indicate that it is a very common species across all forest types in mid-successional stands in interior Alaska.
Figure 5.1. Box and whiskers plots representing the median and quartiles of the total moss cover (%), total lichen cover (%), and the inverse of Simpson’s diversity index in 51 mid-successional stands of interior Alaska according to the forest type. BP: Alaskan birch stands ($n = 5$). PT: Aspen stands ($n = 13$). PM+PT: Black spruce and aspen stands ($n = 6$). PM+BP: Black spruce and Alaskan birch stands ($n = 9$). PM: Black spruce stands ($n = 18$). a) Total moss cover. Different letters indicate a significant difference following a Tukey HSD post-hoc test at $\alpha = 0.05$. Note that the raw data is presented in the graphs, but the analyses were conducted on transformed data to ensure normality and heteroscedasticity. b) Total lichen cover. No significant difference in lichen cover was found among forest types. c) Inverse Simpson’s diversity index calculated on moss and lichen species present in the stands. d) Leaf litter cover.
Figure 5.2. Relationship between leaf litter cover (%) and the square root of the total moss cover (%) in 51 mid-successional stands of interior Alaska according to the forest type. BP: Alaskan birch stands \((n = 5)\). PT: Aspen stands \((n = 13)\). PM+PT: Black spruce and aspen stands \((n = 6)\). PM+BP: Black spruce and Alaskan birch stands \((n = 9)\). PM: Black spruce stands \((n = 18)\). a) Relationship between moss cover and leaf litter cover. b) Relationship between lichen cover and leaf litter cover with the trend line.
Task 6: Tree ring sampling and analysis

The three study regions differed in terms of annual (data not shown) and monthly (Figure 6.2) climate conditions over the thirty-year period examined. Annual climate summaries indicate that the Dalton region was relatively cool and dry, the Steese region warm and moist, and the Taylor region warm and dry. On a monthly basis, the Dalton Highway region experienced a greater change in temperature throughout the year than the other three regions, such that it was relatively warm and dry during the growing season (June, July, and August) compared to the other two regions (Figure 6.2). Regional differences in monthly precipitation were greatest during the growing season, while differences in temperature were greatest during winter.

Standard, averaged chronologies based on site types within regions produced chronology statistics indicating they were suitable for climate analysis. These chronologies demonstrated a negative growth response to previous growing season temperature (July) and current spring temperatures (April and May) at most site types (Figure 6.3). Radial growth responses of trees to precipitation were more variable; trees along the Taylor Highway responded negatively to previous and current May precipitation regardless of site type (Figure 6.4).

The correlation between δ13C chronologies and mean monthly temperatures for a subsmaple of trees in the PFRR indicate that drought stress is likely the mechanism for the observed negative radial growth responses to previous growing season and current spring temperatures. We found that trees on both south and north facing slopes were positively correlated to previous growing season temperature and positively correlated to current season july and august temperature (Figure 6.5). Both south and north facing Δ13C chronologies were positively and negatively correlated to previous growing season and current spring total monthly precipitation, respectively.

The chronologies we developed in this study demonstrate a widespread sensitivity of black spruce radial growth to temperatures in the previous growing season and current spring. These results are in direct contrast to our expectation that black spruce, which are frequently found in the coldest and wettest positions of the boreal forest landscape (Viereck et al. 1983), would respond positively to warm summer temperatures. The most widely accepted explanation for reduced tree growth in response to warmer spring and summer conditions is temperature-induced moisture stress (Barber et al. 2000, Lloyd and Bunn 2007). Warmer temperatures during the previous growing season can result in a reduction in photosynthetic tissue, which limits radial growth the following year (Fritts 1976). Increased spring (April and May) temperatures also restrict radial growth of trees through stimulation of photosynthesis and transpiration of evergreen foliage prior to soil thaw. This can result in severe water stress, as trees are unable to obtain sufficient moisture to sustain their evaporative demands (Berg and Chapin III 1994). In interior Alaska, reduced growth in response to warm spring temperatures has also been observed for white spruce growing at treeline (Ohse et al. 2012) and black spruce trees growing in lowland forests (Wilmking and Myers-Smith 2008). Our preliminary dendroisotopic results indicate that the negative radial growth responses are in fact due to temperature-induced moisture stress.
Figure 6.2. Mean monthly temperature (points) and total monthly precipitation (bars; averaged over the period 1974-2003) for each of the three regions examined in this study. Climate data were obtained from the Scenarios Network for Alaska and Arctic Planning (2013).
Figure 6.3. Correlations between ring width chronologies and mean monthly temperatures. Temperatures were obtained from SNAP climate data over a 17 month climatic window for the Steese (a,d,g,j), Taylor (b,e,h,k), and Dalton (c,f,i,l) Highways. Standard ring width chronologies were developed for south facing (a,b,c), north facing (d,e,f), dry flat (g,h,i) and wet flat (j,k,l) sites. The Y-axis in each figure represents correlation coefficients (black bars indicate significant correlations at p < 0.05), and the X-axis represents months of the year (lower case = year prior to ring formation, uppercase = year of ring formation).
Figure 6.4. Correlations between ring width chronologies and total monthly precipitation. Precipitations were obtained from SNAP climate data over a 17 month climatic window for the Steese (a,d,g,j), Taylor (b,e,h,k), and Dalton (c,f,i,l) Highways. Standard ring width chronologies were developed for south facing (a,b,c), north facing (d,e,f), dry flat (g,h,i) and wet flat (j,k,l) sites. The Y-axis in each figure represents correlation coefficients (black bars indicate significant correlations at p < 0.05), and the X-axis represents months of the year (lower case = year prior to ring formation, uppercase = year of ring formation).
Figure 6.5. Correlations between $\delta^{13}$C chronologies and mean monthly temperature (a,b) and total monthly precipitation (c,d). Climate data were obtained from SNAP climate data over a 17 month climatic window for the Steese region. Standard $\delta^{13}$C chronologies were developed for south facing (a, c), and north facing (b,d) sites. The Y-axis in each figure represents correlation coefficients (black bars indicate significant correlations at $p < 0.05$), and the X-axis represents months of the year (lower case = year prior to ring formation, uppercase = year of ring formation).
Task 7: Monitor invasive species in wildfire and management plots

Fire had a strong influence on non-native plant presence. Non-native plants were never observed in unburned stands, whereas 11 of the 33 surveyed burned stands (eight in the Dalton highway study region, three in the parks highway study region) had non-native plants present ($\chi^2 = 12.32, \text{df} = 1, p < 0.001$). Within burned stands, stand variables were able to correctly classify non-native plant presence in at least 70% of instances. Four of five top models contained residual organic layer depth and/or regenerating paper birch seedlings as model covariates, indicating that they were closely associated with non-native plant presence. Residual organic layer depth was generally lower in sites where non-native plants were present. High densities of regenerating paper birch seedlings were more frequently observed in sites where non-native plants were present.

Seed germination success in the experimental seeding trial in burned forest was affected by substrate type for all species ($p < 0.001$). Germination of the three species was consistently highest on the mineral soil substrate, and no species had successful germination on charred organic soil. Germination on plant litter and regenerating moss was generally lower than observed on mineral soil, although *V. cracca* had higher rates of germination on the litter and moss substrates compared to *M. officinalis* and *T. officinale*. Overall *V. cracca* had the highest total germination (15%) across all substrate types within the burned forest relative to *M officinalis* (7%) and *T. officinale* (4%).

Managed areas

Four of the twelve managed sites had invasive plants present, and none of the undisturbed controls had invasive plants present. Ground cover in the undisturbed control sites was predominantly moss, leaf litter, and lichen, whereas managed sites had a variety of substrates present. *Crepis tectorum* was present in four sites and *Taraxacum officinal* was also present at one of those sites. All of the sites with invasive plants present had been treated by shear blading. Some of the shearbladed sites had exposed mineral soil present, whereas the thinned site did not have mineral soil present. In the thinned site the ground cover was predominantly moss.

Task 8: Provision of data for model development, parameterization, and testing

Among all the descriptors, the landscape category was the best predictor of ROL, accounting for 24.8% of the variability. ROL was lower in flat lowlands ($0.467 \pm 0.177 \text{ cm cm}^{-1}$, mean ± standard deviation) and higher on slopes ($0.712 \pm 0.174 \text{ cm cm}^{-1}$) and flat uplands ($0.605 \pm 0.140 \text{ cm cm}^{-1}$). ROL in flat uplands and slopes were marginally significantly different from each other (Figure 8.2, $p = 0.052$).

To take into account the importance of the landscape categories in predicting ROL, we developed a predictive model of ROL for each of the three landscape categories (Table 8.1). ROL was reduced on sites with a northern exposure for both flat lowlands and slopes. In flat lowlands, ROL was also reduced in sites with high flow accumulation (wetter sites). ROL was
reduced by high OL volumetric water content and increased with the date of burn and the size of the area burned in the models for both flat uplands and slopes. In flat uplands, ROL increased with the slope and the ET/PET ratio.

Our analysis succeeded in reproducing the observed effects of local topography and weather on fire severity. Local topographic effects considered in our analysis included landscape category, slope and aspect, flow accumulation, and the compound topographic index. ROL was lower in flat lowlands characterized by elevated compound topographic index and large flow accumulation than on slopes and flat uplands because moist conditions prevent deep burning of soil organic horizons (Miyanishi and Johnson 2002). Soil moisture is more variable in drier flat uplands and slopes, and it therefore has a larger influence on fire severity in these landscapes compared to lowlands. Among slopes, north facing aspects experience lower evapotranspiration because of lower insolation and higher soil moisture and more protection from deep burning compared to other aspects. The effect of ET/PET of the year prior to fire on ROL in flat uplands suggests that water availability may influence fire intensity over more than a single season by increasing the amount of fuel available through an increase of leaf litter and by reducing soil moisture (Forkel et al. 2012). The positive effect of late-season fire and fire size on ROL is consistent with previous studies that link increase fuel consumption with increased soil drainage and drier soils late in the growing season (Kasischke and Turetsky 2006; Kasischke et al. 2010; Turetsky et al. 2011b).
Table 8.1. Importance and regression coefficient of the parameters selected to predict ROL from the Partial Least Square analysis. N/S is the sine of the aspect quantifying the North/South gradient, ET/PET is the ratio between current and potential evapotranspiration, VWC is the volumetric water content of the OL. The regression coefficients are for the centered scaled data (see Methods).

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Flat lowlands</th>
<th>Flat uplands</th>
<th>Slopes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Importance</td>
<td>Regression</td>
<td>Importance</td>
</tr>
<tr>
<td>Slope (degree)</td>
<td></td>
<td>0.970</td>
<td>0.139</td>
</tr>
<tr>
<td>N/S gradient</td>
<td>0.830</td>
<td>-0.201</td>
<td>0.909</td>
</tr>
<tr>
<td>Flow accumulation (log-transformed)</td>
<td>1.211</td>
<td>-0.334</td>
<td>0.853</td>
</tr>
<tr>
<td>ET/PET (n-1)</td>
<td>0.853</td>
<td>0.108</td>
<td></td>
</tr>
<tr>
<td>VWC</td>
<td>1.287</td>
<td>-0.184</td>
<td>0.882</td>
</tr>
<tr>
<td>Date of burn</td>
<td>1.019</td>
<td>0.146</td>
<td>1.277</td>
</tr>
<tr>
<td>Area burned (km2)</td>
<td>0.893</td>
<td>0.128</td>
<td>0.875</td>
</tr>
</tbody>
</table>
Figure 8.2. ROL for different landscape categories based on field observations (FL = flat lowland, FU = flat upland and S = slopes). The length of the box represents the inter-quartile range, the dot and the horizontal line in the box represents the group mean and median, respectively. The vertical lines extend to the minimum and maximum values. The different letters below the boxes indicate significant differences in ROL among landscape categories.
Task 9: Forecast future landscape distribution with coupled models

TEM Development and Testing

The three fire severity models we developed for each landscape category explained 49.6% of the observed variability of ROL (Figure 9.2). There was no significant difference between observed and simulated ROL (paired t-test: t = 0.01, p = 0.992) and the effect of landform categories on ROL was faithfully reproduced (F= 16.82, p < 0.001, Figure 9.2). The parent version of DOS-TEM, which used a look-up table approach, significantly underestimated observed ROL (paired t-test: t = 7.67, p < 0.001) and the difference of ROL between landform categories was not significant (F = 0.13, p = 0.878; Figure 9.2). The effect of landform categories and fire on the OL thickness impacted the active layer thickness. In general, the relative organic layer loss after fire induced an increase of the relative active layer thickness, defined as the increase of active layer thickness (ALT) after fire divided by the pre-fire ALT (F = 52.82, p < 0.001; Figure 9.3). The relationship is significant in flat uplands (F = 16.40, p < 0.001) and slopes (F = 37.95, p < 0.001), but not in flat lowlands (F = 2.61, p = 0.135).

DOS-TEM simulated changes in OL thickness, active layer thickness, and soil C stocks for the sampled sites between [1901-1930] and [2071-2100] across landscape categories and the factorial warming and fire scenarios (Table 9.1). The dynamics of OL thickness, active layer thickness, and soil C pools were significantly influenced by fire regime and warming. While the effect of topography was significant on the dynamic of OL and active layer thicknesses; it was only marginally significant (p = 0.06) on changes in soil C pools. There were marginally significant and significant interactions between local topography and warming effects on changes in OL thickness and ALT respectively, suggesting a weaker effect of warming in flat lowlands compared to flat uplands and slopes. Warming decreased OL thickness by 2.46, 4.19 and 4.98 cm and increased ALT by 46.3, 76.3 and 84.6 cm for flat lowlands, flat uplands and slopes, respectively. The interaction between topography and fire severity was significantly related to OL thickness (Figure 8.2), and soil C stocks (marginally significant, p = 0.06), with a general pattern of weaker effects of fire in flat lowlands compared to flat uplands and slopes. The interaction between fire regime and warming was significant for changes in OL thickness and soil C stocks, suggesting that the effect of fire was intensified with warming climate. This interaction, which was significant at the plot level (Table 9.1), was reduced at the regional level (Figure 9.4), because warming affected the entire landscape while only 44.7% of the landscape was estimated to have burned between 2010 and 2100.

Our simulation for historical warming and change in fire regime over the entire study area (Technical approach: Figure 8.1) indicates that during the course of the 20\textsuperscript{th} and 21\textsuperscript{st} centuries, the OL thickness decreased, active layer thickness increased, and soil C decreased (Figure 9.4). Warming alone caused a decrease of 5.4 cm in OL thickness and 7579 g C m\textsuperscript{-2} soil organic C loss by 2100. The addition of changes in fire regime decreased OL thickness by 6.5 cm and soil C stocks by 9817 g C m\textsuperscript{-2} by 2100 compared to the simulation with no change in fire and no warming. Changes in fire regime only induced a loss of 8792 g C m\textsuperscript{-2} of soil C stocks by 2100. This loss was due to 1102 g C m\textsuperscript{-2} from soil C combustion and 7690 g C m\textsuperscript{-2} from soil C decomposition (Figure 9.5). The increase in active layer thickness by 2100 was similar (about 1.1 m) in response to warming alone, changes in fire regime alone, and the combination of
warming and changes in fire regime. However, the change in active layer thickness from the warming alone had a different trajectory than the other two scenarios, indicating that changes in fire regime exposed more soil C to decomposition during the 21st Century than did warming.

ALFRESCO Development and Testing

Preliminary results of the ALFRESCO calibration and testing of historical fire regime trends has been completed. Comparison of model simulation results with observed (1950-2010) wildfire activity indicated good agreement for both cumulative area burned and the cumulative distribution of individual fire size. Simulated cumulative area burned for the historical period 1950-2010 showed good agreement with observed trends (Figure 9.6). Individual simulation replicates (n = 200) vary closely around observed fire activity and the best performing model replicates match closely to observed cumulative area burned in 2010. All model replicates underestimate the record wildfire activity of 2004 and 2005. The distribution of individual simulated fire sizes for the period 1950-2010 also matches well with the observed record (Figure 9.7). Both simulations and historical observations indicate a large proportion of cumulative area burned is the result of a relatively small number of very large individual fires. The interannual variability of annual total area burned was simulated adequately by the model (Figure 9.8). The model currently and consistently underestimates annual area burned in the years with the largest observed fire activity and specifically the years 2004 and 2005. The spatial extent of wildfire occurrence matches well with general observations of fire activity (Figure 9.9). The great majority of wildfire activity occurs in the interior boreal forest region of Alaska. This region is bounded by the Brooks Range to the North (and tundra ecosystem types) and the Alaska Range to the south (and substantial increases in monthly and annual precipitation).
Table 9.1. Results of the analysis of variance testing the effect of local topography, warming and fire intensification on changes in averaged OL thickness, active layer thickness, soil carbon stocks from [1901-1930] and [2071-2100] in the sites where observations have been collected. DF = degree of freedom, F = Fisher value, p = probability.

<table>
<thead>
<tr>
<th>Source</th>
<th>DF</th>
<th>Change in OL thickness</th>
<th>F</th>
<th>P</th>
<th>Change in ALT</th>
<th>F</th>
<th>P</th>
<th>Change in Soil carbon stock</th>
<th>F</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>Local topography</td>
<td>2</td>
<td>9.80</td>
<td>21.05</td>
<td>&lt;0.01</td>
<td>2.98</td>
<td>0.06</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Warming</td>
<td>1</td>
<td>19.54</td>
<td>97.08</td>
<td>&lt;0.01</td>
<td>15.81</td>
<td>&lt;0.01</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Local topography*Warming</td>
<td>2</td>
<td>2.58</td>
<td>4.38</td>
<td>0.01</td>
<td>0.28</td>
<td>0.76</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fire</td>
<td>1</td>
<td>359.76</td>
<td>6.4</td>
<td>0.01</td>
<td>272.65</td>
<td>&lt;0.01</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Local topography*Fire</td>
<td>2</td>
<td>35.27</td>
<td>1.90</td>
<td>0.16</td>
<td>2.88</td>
<td>0.06</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Warming*Fire</td>
<td>1</td>
<td>12.62</td>
<td>1.13</td>
<td>0.29</td>
<td>3.80</td>
<td>0.05</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Local topography<em>Warming</em>Fire</td>
<td>2</td>
<td>0.16</td>
<td>0.08</td>
<td>0.93</td>
<td>0.07</td>
<td>0.93</td>
<td></td>
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</tbody>
</table>
Figure 9.2 Model verification: comparison of the ROL computed from observations and from simulated by DOS-TEM with the original version of the model (closed circle) and with the modified fire module (open circle). The vertical lines represent standard deviation of the simulated ROL, the horizontal line represent standard deviation of the observed ROL for each landscape categories. The oblique line represents the [1:1] slope.
Figure 9.3. Relation between relative gain of ALT and relative loss of organic layer in response to fire.
Figure 9.4. Time series of changes in a) organic layer thickness, b) active layer thickness and c) soil carbon content from 1901 to 2100 for the equilibrium scenario (grey solid line), warming only scenario (grey dotted line), for the fire intensification only scenario (black dotted line), for the warming and fire intensification scenario (black solid line).
Figure 9.5. Amount of soil carbon lost from fire only partitioned between decomposition of organic matter (solid black line) vs. combustion during fire (dotted black line).
Figure 9.6. Cumulative area burned (km$^2$) through time. Record of observation 1950-2010 indicated by the red line. Individual model simulations (n = 200) indicated by gray lines. Projections through 2099 are for the CCMA A1B climate scenario.

Figure 9.7. Cumulative area burned versus individual fire size (km$^2$) for the historical observation period 1950-2010. Red line indicates observed distribution. Green and blue lines indicate best performing simulation replicates. Gray lines indicate individual replicates (n = 200).
Figure 9.8. Annual area burned (km$^2$) for the historical observation period 1950-2010. Black bars indicate historical observations and red bars indicated simulated results from best performing individual model replicate ($n = 1$).
Figure 9.9. Map showing simulated fire scars for the retrospective period 1900-2010. Individual fire scar colors indicate age of burn going from oldest (red) to youngest (green).
Task 10: Planning and participation in workshops for Alaska land managers

Question 1: How does simulated fire frequency respond to different climate scenarios during the 21st Century on, and adjacent to, military lands of the Upper Tanana Valley?

Strategy: Analyze simulations for the Upper Tanana Hydrologic Basin (upstream of the Nenana River Sub-basin) and for military lands within the Upper Tanana Hydrologic Basin.

Analyses of Interest: (1) Chronology of percent of land annually burned within the analysis units (areas of interest) through the 21st Century (this may need to be summarized on a decadal scale to reveal long term trends); (2) Maps of probability of burning during the 21st Century for analysis units at 1-km resolution (for purposes of identifying hotspots).

Question 2: How might changes in the fire management options within military training land boundaries influence the frequency and extent of wildfire activity on, and adjacent to, military lands in the Upper Tanana Valley during the 21st Century?

Strategy: Explore hypothetical changes to the spatial boundaries of current fire management options. For example, explore model response to a conversion of current limited suppression zones to full suppression zones. Develop a suite of experiment simulations and analyze the variables of interest in questions 1 and 2 and compare with the results from those earlier simulations.

Analyses of Interest: (1) Chronology of percent of land annually burned within the analysis units (areas of interest) through the 21st Century (this may need to be summarized on a decadal scale to reveal long term trends); (2) Maps of probability of burning during the 21st Century for analysis units at 1-km resolution (for purposes of identifying hotspots).

Question 3: How might wildlife (e.g. moose) habitat suitability change on military lands in the Upper Tanana Hydrologic Basin through the 21st Century?

Strategy: Analyze Integrated Ecosystem Model (IEM) wide simulations for military lands within the Upper Tanana Hydrologic Basin.

Analyses of Interest: (1) species specific habitat suitability on military lands in the Upper Tanana Hydrologic Basin through the 21st Century (this may need to be summarized on a decadal scale); (2) Maps for analysis at 1-km resolution of vegetation composition and age structure during the 21st Century. Species of interest and vegetation age thresholds established collaboratively with DoD land managers would be developed to characterize changes in habitat composition and extent.
Conclusions to Date

Task 3. Fire management practices, especially shearblading dramatically reduce ecosystem C storage and permafrost stability, and alter forest composition. Analysis of field data is ongoing and will include more detailed investigation of differences across study sites and treatment types, as well as further study of the relationship between changes in the soil organic layer and permafrost thaw.

Task 4. These results indicate that there are large differences in C and nutrient distribution and cycling in Alaska paper birch and black spruce forests. More C is stored aboveground in wood in birch stands while spruce stores much more C in the soil organic layer. While the birch stands exhibit larger annual foliar litter inputs and higher nutrient concentrations in that litter, the significantly smaller organic layer in these stands suggests that nutrient turnover is occurring more rapidly in birch stands. This is consistent with our N mineralization and resin data. Ongoing analyses include calculation of soil CO2 fluxes from a laboratory experiment, estimates of net primary productivity, and project synthesis. Continued field measurements of leaf litter inputs, soil temperature, resin N and P accumulation, and mass-loss decomposition will continue for the remainder of this SERDP project.

The preliminary results of the moss transplant experiment are consistent with our expectation that the effect of leaf litter addition on moss growth will take some time before it may be detected. Since it has only been one year since the litter addition was first applied on the transplants, we would expect that treatment does not yet have a clear effect on moss growth. Moreover, we observed a 23% decrease in leaf litter cover on transplants with the addition treatment between October 2012 and June 2013, which might have mitigated the impact on moss. This issue was addressed in 2013 by adding netting on the litter addition to make sure that it would remain in place. Interestingly, there seemed to be a difference in size for the new shoots (2013) between transplants in birch and spruce stands, but this difference was not observed in one year old shoots (2012). That may suggest that moss growth might occur earlier in birch stands (late summer - fall) than in spruce stands (following spring), but more investigation is necessary.

Future work will be required to assess the impact of leaf litter on moss growth. The leaf litter manipulation and moss measurements will be continued in 2014 and 2015, through in-kind support from the Bonanza Creek Long Term Ecological Research Program. We will also complete the steps necessary to build allometric equations that will allow us to estimate biomass accumulation in the transplants. Statistical analyses comparing biomass accumulation according to treatment and forest type will be conducted. Soil temperature under the transplants will continue to be recorded in 2014 and 2015, to maximize the duration of the temperature record.

Task 5. Our preliminary assessment of the variation in moss communities and functional traits across deciduous, mixed and coniferous forest types suggests that there are differences in the moss patterns among forest types. More detailed analyses will be pursued in 2014 in order to clarify the relationships between moss species, stand types, soil types, topography, and leaf litter cover.
We are planning to expand the analysis to include both late (more than 62 years since fire) and early successional (less than 20 years since fire) stands in order to determine a critical period for moss and lichen establishment. Field sampling for early successional stands will be conducted in the summer of 2014, and completion of the analyses is planned for the end of 2014.

**Task 6.** Our assessment of dendroclimatic responses within interior Alaska suggest that the effects of continued warming on black spruce tree growth will largely be negative. Preliminary results from our dendroisotopic analysis indicate that the mechanism for reduced radial growth in response to previous growing season and current spring temperatures is temperature-induced moisture stress. However, these analyses were completed on a small samples of trees, with considerable individualistic variability, thus we plan to sample an additional 24 trees, 12 on each the south and north facing slopes, to ensure the robustness of our results.

The observed negative radial growth response to continued warming is likely to be widespread, occurring at all landscape positions and in all regions. As black spruce is one of the most widespread forest types in the boreal forest, understanding its response to continued warming is of particular importance and drought stress related mortality could lead to the elimination of negatively responding trees. Additionally, trees with negative responses to warmer and drier growing conditions could be an indicator of decreased ecosystem resilience (Johnstone et al. 2010). Thus, with disturbance from fire, areas dominated by drought stressed trees would be less likely to recover to pre-fire stand densities. As data on seedling regeneration was collected within each site, we plan to examine the possible linkages between growth-climate responses and ecosystem resilience. The change in pre-fire to post-fire stand composition will be used as a proxy for ecosystem resilience, with sites experiencing the largest change in composition indicative of decreased resilience and an altered successional trajectory. We will likely use a mixed-modelling approach to predict the change in pre-fire to post-fire density based on the dendroclimatic responses and environmental variables.

**Task 7.** This study provides evidence that fire disturbance and forest management facilitates non-native plant colonization in black spruce forests in interior Alaska. Non-native plants were observed in recently burned and managed areas, but not in mature forest stands adjacent close to roadside seed sources, indicating that mature forests are resistant to non-native plant colonization under current conditions. Legacy effects of the thick organic layer found in mature forest stands appears to influence site invasibility after fire. We found that residual organic layer depth was negatively associated with the presence of non-native plants in black spruce forests that had recently burned. Furthermore, we observed no successful seed germination in our seeding trials on charred, intact organic soil, in contrast to mineral soil substrates. Thick residual organic material remaining after fire creates an unfavourable substrate for germination and growth due to the high porosity and dark colour of the substrate, which lead to high temperatures and low moisture content that inhibit germination and cause seedling desiccation (Johnstone and Chapin, 2006). In contrast, shallow residual organic layers and exposed mineral soil provide favorable sites for germination and greatly increase the potential for tree seedling recruitment (Johnstone and Chapin, 2006). We found that the abundance of paper birch seedlings was a strong predictor of non-native plant presence. Paper birch is a small seeded species with few carbohydrate reserves, and favors exposed mineral soils over residual organic soils (Johnstone and Chapin, 2006). Thus, birch seedling abundance may be a proxy for increased seedbed quality.
Task 8. We have organized and analyzed the data on the consumption of soil organic matter in interior Alaska, and we have successfully developed a predictive model of relative organic layer loss based on topographic and other variables. In task 9 below we present further analysis after incorporating this predictive model into a process-based ecosystem model. As indicated above, we are currently organizing information from the project on vegetation succession and the re-accumulation of the vegetation and soil organic C following fire for use in model development, parameterization and testing.

(1) TEM fire severity module has been developed from data collected in black spruce burns exclusively. Combustion rates and site characteristics collected in deciduous and mixed stands for Task 2 are being used to develop our fire severity module and extend its application to these vegetation communities.

(2) Data collected on seedling composition, survival and growth along with quantification of fire severity, and site characteristics measured in black spruce stands in Task 1 are being used to parameterize a recruitment model. This recruitment model determines the trajectory of the vegetation succession after fire, among an evergreen-dominated, a deciduous-dominated and a mixed trajectory (i.e. an early deciduous-dominated stage succeeding to a late evergreen-dominated stand), as a function of fire severity, pre-fire stand characteristics and drainage conditions.

(3) Natural seedling and forest stand biomass and growth rates measured in early (Task 1) and mid-successional (Task 5) stages respectively are being used to validate TEM simulations of dynamic of vegetation biomass after fire, for evergreen-dominated and deciduous-dominated trajectories.

(4) Detailed data collection on vegetation and soil C and N pools, soil physics and decomposition rates conducted in the Murphy Dome mid-succession sites (Task 5) are being used do assess the parameters related to the soil and various plant functional simulated by the Dynamic Vegetation and Organic Soil Models of TEM (DVM-DOS-TEM) for black spruce and birch vegetation communities.

(5) The measurement of active layer, soil temperature and organic layer thicknesses collected in Task 1 and 2 are and will be used to validate TEM simulations of the effect of the organic layer on permafrost stability.

Task 9: We have successfully tested and evaluated TEM with respect to its ability to simulate the loss of organic matter after fire, and a manuscript has been published describing this research (Genet et al. 2013). We have also successfully tested and evaluated ALFRESCO with respect to its ability to simulate the fire regime in interior Alaska in terms of historical patterns of area burned and fire size distribution. We are currently conducting an asynchronous simulation for interior Alaska in which the fire simulated by ALFRESCO is being used to drive TEM for historical climate and for future scenarios of climate change. We anticipate having these simulations analyzed by the end of June 2014. During the summer, we plan to finish the synchronous coupling of the models in which the simulation of organic layer loss by TEM will influence the simulation of fire by ALFRESCO. We will conduct the simulations with the synchronously coupled models in fall 2014 with results analyzed and compared with the asynchronous simulations by the end of 2014.
Task 10. We will continue to meet with the fire and land managers through the remainder of the project to refine the information products we provide to them. A finalized set of information products will be delivered to the fire and land managers by the end of 2015.


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