Final Wildlife Research Report, ADF&G/DWC/WRR-2015-1

Identifying and evaluating techniques for wildlife habitat enhancement in Interior Alaska: Prescribed burn assessment

Thomas F. Paragi
Dale A. Haggstrom

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Alaska Department of Fish and Game Division of Wildlife Conservation

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Thomas F. Paragi
Wildlife Research Biologist
1300 College Road
Fairbanks, AK 99701-1551
tom.paragi@alaska.gov

Dale A. Haggstrom[1]
Wildlife Biologist
1300 College Road
Fairbanks, AK 99701-1551

Alaska Department of Fish and Game
Division of Wildlife Conservation
1300 College Road
Fairbanks, AK 99701-1551

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[1] Retired in 2010.

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Cover Photo: Aerial ignition pattern of prescribed fire on the Mosquito Flats burn northwest of Tok, Alaska.

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Abstract

We sought to evaluate the initial efficacy of a July 1998 landscape-scale (214 km^2) prescribed burn in the East Fork Dennison Fork of the Fortymile River to convert spruce (*Picea* spp.)-dominated stands to grasses, forbs, shrubs, and hardwood saplings beneficial to early-seral wildlife in eastern Interior Alaska. We used 1:63,360-scale color-infrared aerial photos from 1981 and 1983 and Quickbird satellite imagery (resolution: 2.6 m multispectral, 0.6 m panchromatic; DigitalGlobe™, Longmont, Colorado; presently distributed by Satellite Imaging Corporation, Houston, Texas, http://www.satimagingcorp.com) from July 2002 to estimate vegetative type conversion after 4 growing seasons in a 25 km^2 portion of the burn. A normalized difference vegetative index (NDVI) applied to a 272 km^2 Quickbird scene from September 2002 that included most of the burn allowed us to revise the 1998 helicopter mapping of the burn perimeter and estimate that fire occurred on 89% of the area within the perimeter. A supervised object-oriented classification of polygons was performed on pre- and post-fire images using eCognition® software (Definiens® Imaging, Munich, Germany; presently distributed by Trimble® Geospatial Imaging, Sunnyvale, California, http://www.ecognition.com). The pre-burn classification of 80% conifer, 15% shrub, and 1% grass changed to 7% conifer, 53% shrub, and 38% grass by 4 growing seasons after the fire, with shrub class dominated by aspen (*Populus temuloides*) saplings, willow (*Salix* spp.), and some alder (*Alnus* spp.). Several factors hindered acquiring a large number of ground validation points for building a classification error matrix. We did not attempt to estimate and validate a spatial layer on fire severity (an important factor in initial post-fire vegetative response) 4 years post-hoc but used a digital elevation model to derive terrain features and focused analysis on conversion of conifer to shrub only. The large number of pixels (~17 million) with the high resolution imagery affected computational time. We split the 25 km^2 study area into northern and southern sections for both computational considerations and to guard against misinterpreting spurious results by independently analyzing each half and comparing results. We modeled terrain effects on vegetation conversion after fire with nonspatial and spatial generalized linear models. We used a low rank radial smoother to model the spatial correlations, and our final spatial model included the covariates elevation and slope and their interaction. Our modeling of seral transition from woodland spruce to shrub and deciduous sapling indicated a preference for transitioning to shrub rather than grass at lower elevations and steeper slopes for all but extreme (≤5% of area) elevation and slope combinations found in the study area. Not having fire severity as a spatial variable likely limited the predictive capability of our model and precluded a mechanistic understanding of regeneration potential, so caution is warranted in applying our results on terrain effects in future planning of prescribed burns. Recommendations are given for future evaluation of vegetative and wildlife responses to prescribed burns.

Key words: Aspen, birch, boreal forest, change detection, fire severity, imagery, moose browse, object-oriented classification, prescribed fire, spruce, willow.

Introduction

The boreal forest of Interior Alaska is primarily open canopy, slow-growing black spruce (*Picea mariana*) interspersed with occasional dense, well-developed stands and treeless wetlands and stands of white spruce (*P. glauca*) or deciduous trees such as quaking aspen (*Populus tremuloides*) or Alaska birch (*Betula neoalaskana*) in warmer floodplains and south-facing slopes (Viereck 1973, Calef et al. 2005). Wildland fire is the major disturbance that influences vegetative succession, forest composition, and ecological processes on boreal upland sites (Viereck 1973, Foote 1983, Johnson 1992). Fire can remove the forest canopy and the organic duff layer of the forest floor. With increased sunlight and decreased duff insulation, the soil typically warms and allows thawing on permafrost sites, which accelerates nutrient cycling after a burn (Chapin et al. 2006b).

Although Native peoples historically used fire to maintain openings in the forest for hunting and other purposes across northern Canada (Lewis and Ferguson 1988) and Alaska (Roessler 1997, Natcher et al. 2007), concerns over human-caused and lightning-caused fires increased starting in the 1890s with the gold rush and influx of European inhabitants to Alaska (Todd and Jewkes 2006). Fire suppression began near settlements and spread further into wildlands in the 1950s with aircraft-aided detection and delivery of resources (Todd and Jewkes 2006). However, the attempt to suppress all fires over such a large area greatly increased suppression costs and, during periods of high fire activity, exhausted the limited supply of fire resources. Concurrently, land and resource managers began to recognize the important ecological role of fire in the boreal forest and the negative consequences of attempted fire exclusion, such as contiguous fuels near communities or desired resources like timber. These disparate needs for change in fire policy led to the development of interagency fire management plans in the 1970s and 1980s. Fire management plans allowed land managers to select predetermined initial response strategies based on the values identified for protection and natural resource management objectives (Alaska Wildland Fire Coordinating Group 1998).

In the mosaic of stand types and ages in boreal forest, conifer food webs are often characterized by invertebrates and their avian predators, whereas young deciduous or broadleaf forest is more generally the forage base for mammalian food webs (Pastor et al. 1996). Species diversity and the abundance and productivity of wildlife are often positively correlated to recent disturbance in boreal forest that creates early-successional habitats (Gullion 1984, Haggstrom and Kelleyhouse 1996). This need for periodic site disturbance is greatest in areas where the public desires increased opportunities to hunt and view wildlife, such as with intensive management (Title 16, Alaska Statutes 16.05.255[e]-[g] and [k], http://www.legis.state.ak.us/basis/statutes.asp#16.05.251). In these areas, particularly those which are road-accessible and situated near communities, active habitat management may be required (Haggstrom and Kelleyhouse 1996). Habitat enhancement must be effective and efficient to be affordable at the landscape-scale necessary to achieve wildlife abundance and harvest objectives (e.g., Title 5, Alaska Administrative Code 92.108), and must be done in an acceptable manner. Public expectations for the management of wildlife and public land may conflict, such as where high herbivore abundance desirable to hunters may hinder production of timber (Andrews 1998, Angelstam et al. 2000). Managers conducting landscape-scale prescribed burns have an obligation to objectively evaluate whether the treatment achieves the stated goals in a cost-effective manner, so the public can decide whether the results justify the expense and risk (e.g., smoke drifting into communities, causing health concerns, or delays in air traffic).

When fire is used to rejuvenate existing vegetation or change cover type, the burn prescription is the set of weather parameters (wind, relative humidity, fuel moisture conditions) under which a given fuel type is ignited. Burn prescriptions describe the desired fire behavior, which includes flame length, rate of spread, fire intensity (heat released per length of front; Taylor et al. 1996), and fire severity, which is the near-term effects on succession based on the extent to which surface materials and the organic layer is consumed. Low fire severity tends to favor sprouting from a surviving root system in hardwoods and self-persistence in conifers, whereas high fire severity tends to favor seed germination for early establishment of hardwoods and shrubs on mineral soil (Viereck 1973, Johnstone and Kasischke 2005). We follow the terminology of Lentile et al. (2006) to distinguish fire severity from burn severity, which more broadly describes longer-term ecological effects of fire.

Game management programs to increase moose (*Alces alces*) harvest may require habitat enhancement in specific areas to improve forage production, particularly when populations exist at high density (Boertje et al. 2007). Planning of landscape-scale prescribed burns to enhance moose winter range requires choosing areas within a range of upland and floodplain habitats where fire can be safely and effectively applied to maintain or increase stem density of browse species 1.5–3.0 m tall (Seaton 2002). Aside from safety considerations, the area proposed for prescribed burns is primarily based on the flammability of vegetation types and likelihood for seral conversion. The Alaska Department of Fish and Game (ADF&G) has a fire policy that encourages wildland and prescribed fire management practices and decisions that benefit the fish and wildlife resources of Alaska. The department participates in the Alaska Wildland Fire Coordinating Group (http://fire.ak.blm.gov/administration/awfcg.php). Since the mid-1990s, ADF&G's Division of Wildlife Conservation (DWC) has received dedicated funds from the Alaska legislature for enhancement and restoration of wildlife habitat by means of prescribed fire and mechanical treatments. Whereas stand-scale burns <12 hectares have been relatively expensive because of firefighter labor and other fixed costs ($790/ha; Paragi and Haggstrom 2007), 3 landscape-scale burns composing 36,240 ha conducted by aerial ignition from helicopters were comparatively inexpensive, averaging $0.84/ha (Haggstrom 1999). The large prescribed fires included the East Fork burn in 1998 and Kechumstuk and Mosquito Fork burns in 1999, all in Game Management Unit 20E in eastern Interior Alaska.

The primary management goal of the prescribed burn in the East Fork Dennison Fork of the Fortymile River (East Fork burn) was to produce a combination of severe and moderate burn intensities to restore stand age diversity among vegetative types that would benefit wildlife species needing early to mid-successional habitat (Kraemer and Haggstrom 1998). This burn is the subject of our study. Gardner (1998) described moose management and habitat objectives that included a desire to maintain a natural fire regime and conduct prescribed burns to benefit moose. A secondary management goal of the East Fork burn was to reduce the continuity of crown fuels, thus reduce the risk of unmanageable, expensive, and potentially dangerous wildland fires that could threaten resources on adjacent lands.

A maximum area (allowable) perimeter encompassing 158,000 ha defined the area within which the prescribed fire could spread without need for suppression action (Kraemer and Haggstrom

1998). Four ignition ("burn") units totaling 34,290 ha were delineated within a maximum area (allowable) perimeter (Fig. 1). The management objectives of the burn (Kraemer and Haggstrom 1998:4–5) were to "(1) treat 50–70% of each unit under weather and fuel moisture conditions where duff removal will range from moderate to maximum; (2) kill >50% of the black spruce occurring in the final burned area with a burn of varying intensities where duff removal is close to mineral soil to allow shrub understory component to proliferate by seeding; and (3) kill >50% of the above ground stems of black spruce, aspen, poplar and willow occurring in the final burn with less fire intensity to promote root or basal sprouting." The plan allowed burning between 1 June and 30 September.

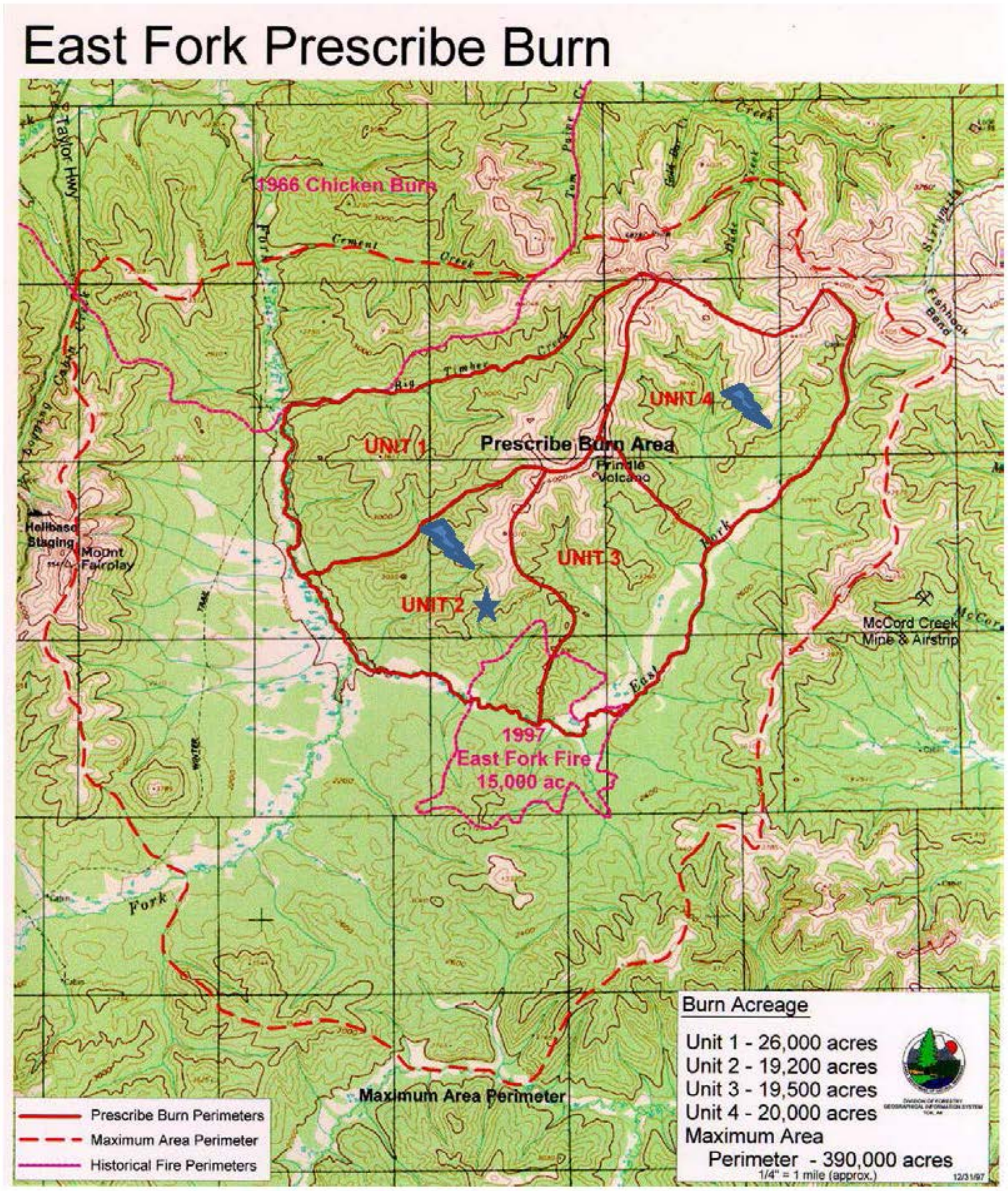


Figure 1. Topographic image from the East Fork burn plan showing a maximum area (allowable) perimeter, 4 ignition units designated for operations planning by fire professionals, and wildland fire history (Kraemer and Haggstrom 1998:Appendix G). Approximate locations of a remote automated weather station (star) and prescribed ignition areas (lightning bolts, Units 2 and 4) on 21 July 1998 were overlaid based on a drawing by R. Kraemer, Alaska Division of Forestry, Tok.

The 1998 East Fork burn was the first landscape-scale prescribed burn in Alaska to our knowledge. When the lead author was hired in 1999 to conduct research on efficacy of habitat enhancement techniques, we sought to evaluate the success of the East Fork prescribed burn in converting spruce-dominated stands to early-successional forbs, shrubs, or hardwood saplings. The study began under federal aid project 5.0 as objective 8 under job 3 (Paragi and Haggstrom 2005) and continued under project 5.10 as job 1e (Paragi et al. 2009). Our goal was to determine which site factors in the 1998 East Fork burn produced a desirable outcome of successional type conversion from conifer to shrub that would enhance moose browse. This information could aid planning of future prescribed burns. Our research objectives were to determine the extent of burned area; determine vegetation type change by comparing pre-burn aerial photography to post-burn photography and imagery; and determine the influence of elevation, slope, and aspect on vegetation type change.

Study Area

The prescribed burn was conducted in the Fortymile River drainage of eastern Interior Alaska about 75 km northeast of Tok and 40 km west of the Yukon border (63°43′N, 141°52′W; Fig. 1). The area is uninhabited public land, and developed access is minimal. A maximum area (allowable) perimeter was centered on part of a ridge system oriented roughly northeast from the confluence of the East Fork Dennison Fork and the Dennison Fork rivers. Terrain elevation varies from the river confluence at 615 m through rolling hills and steep ravines to rocky peaks at 1,410 m.

Vegetation in the burn area was typical of Alaska boreal forest and varied with elevation, aspect, and drainage (Viereck et al. 1992, Chapin et al. 2006a). Expanses of black spruce with tamarack (*Larix laricina*) and Alaska birch dominated cooler, wetter sites, whereas white spruce, quaking aspen, and balsam poplar (*P. balsamifera*) occupied warmer or drier sites. Aspen was generally more common than Alaska birch. Substantial areas were black spruce-needleleaf woodland with 10–25% canopy cover (Viereck et al. 1992:20–21). Mosaics of shrub and herbaceous communities occurred among forested areas. Tree line occurred at about 1,400 m with alder (*Alnus* spp.), bog birch (*B. glandulosa*), dwarf birch (*B. nana*), and willow (*Salix* spp.) typical in the transition zone from forest to alpine tundra. Willow and birch shrubs often occurred in mixed stands and were more common than alder, which occurred singly or in small patches with black spruce. Mature black spruce stands in the western lowland portion of the prescribed burn area were 66- to 230-years old (Kyle Joly, U.S. Geological Survey, in litt., 2004).

There have been no soil surveys specific to the study area, but soil associations with vegetation are described for the physiographic region "Interior Alaska highlands" by Rieger et al. (1979). The area is underlain by discontinuous permafrost (Washburn 1973:Fig. 3.6). We did not measure depth to permafrost but presumed it to be generally shallow except for birch-aspen-white spruce forest on south-facing slopes (Rieger et al. 1979:10, 93).

The burn prescription (Kraemer and Haggstrom 1998) was prepared using the Canadian Forest Fire Danger Rating System and the 1993 Canadian Fire Behavior Prediction System as tools to track forest fuel moistures, predict spread rates, and estimate burn intensities (Taylor et al. 1996). The prescription was intended to allow ignition under a wide range of midsummer burning conditions while avoiding the extremes: conditions so marginal that only a light surface burn is obtained and conditions so flammable that a very severe burn is obtained over most of the area.

After 8 days without precipitation, ignition was initiated within prescription parameters on 21 July 1998 in 2 of the 4 planning units (Fig. 1) in advance of an approaching weather front forecast to bring rain to the area. A helicopter was used to distribute plastic spheres containing potassium permanganate that was activated by injection of ethylene glycol to produce a delayed exothermic reaction. Observations at the time of ignition were relative humidity 33%, wind 18 km/hr generally from the East, Duff Moisture Code 49.5, and Drought Code 246. Calculated parameters were Initial Spread Index 9.7, Buildup Index 67.3, and Fire Weather Index 25.2. Most burning occurred within a few hours of ignition. Relative humidity increased throughout the day, creating less favorable burn conditions as the day progressed. By the morning of 22 July, the relative humidity was too high to continue ignition, and the area received 14.7 mm of precipitation during the day. Observations by fire managers mapping the burn perimeter from the helicopter on 21 July (Fig. 2) suggested that the desired mosaic of unburned areas interspersed among burned areas of varying severity had been achieved within the 21,440 ha burn area (Fig. 3). An aerial observer on 5 August noted numerous points of smoke on the burn perimeter and within its interior, with fire visible only on the northwest corner (Ray Kraemer 1998, Division of Forestry [DOF] files, Tok). The cost for the fire specialists to plan and implement this burn was $0.74/ha ($0.30/ac), excluding ADF&G's Division of Wildlife Conservation (DWC) staff time to assist with planning.

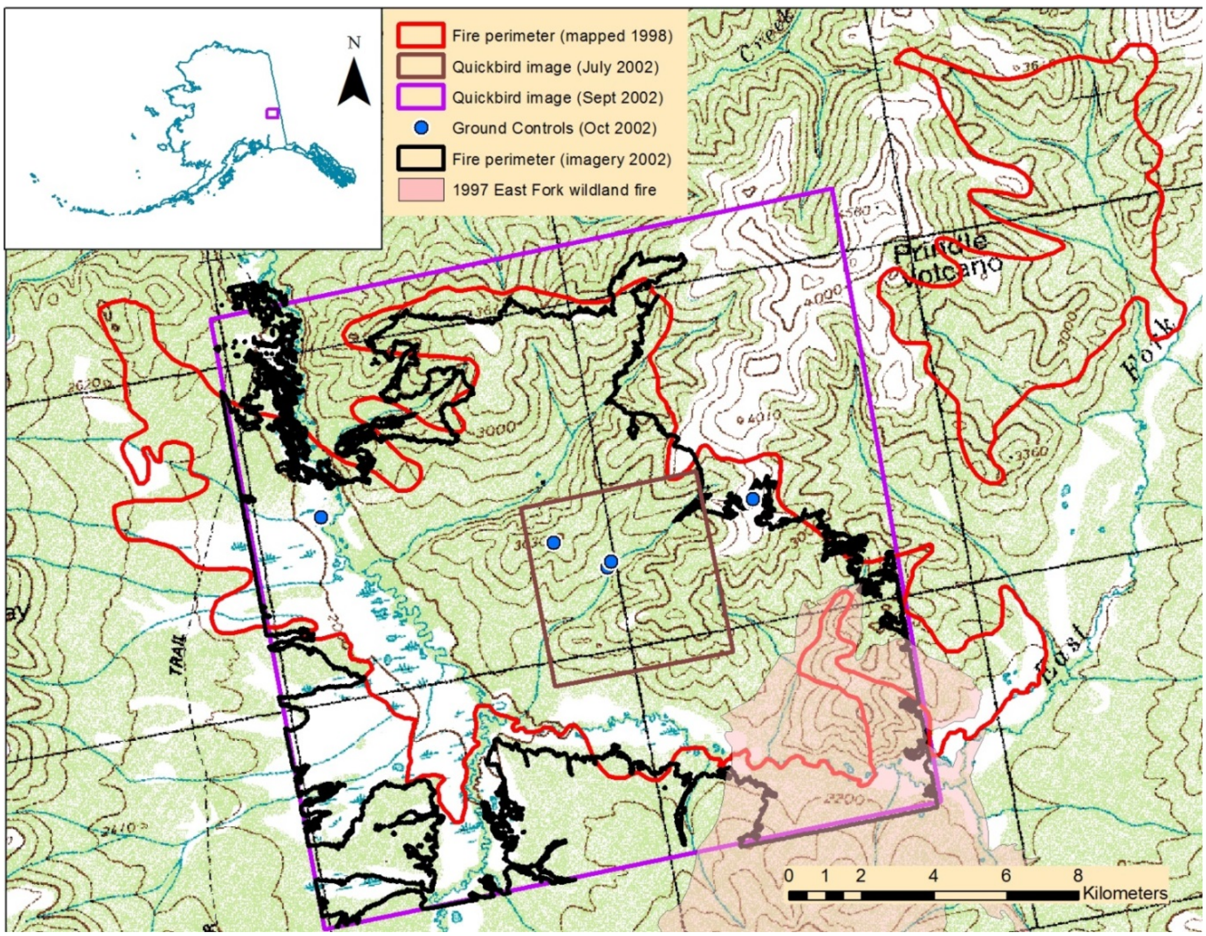


Figure 2. Location of both perimeters of the 1998 East Fork prescribed burn and associated satellite imagery acquired in 2002 to evaluate the effects of the 21 July 1998 prescribed fire on vegetation in Game Management Unit 20E, eastern Interior Alaska. The 25 km^2 Quickbird image from July 2002 defined the study area for analysis of change detection between pre- and post-burn vegetation. Ground control points to orthorectify the September 2002 image are shown.

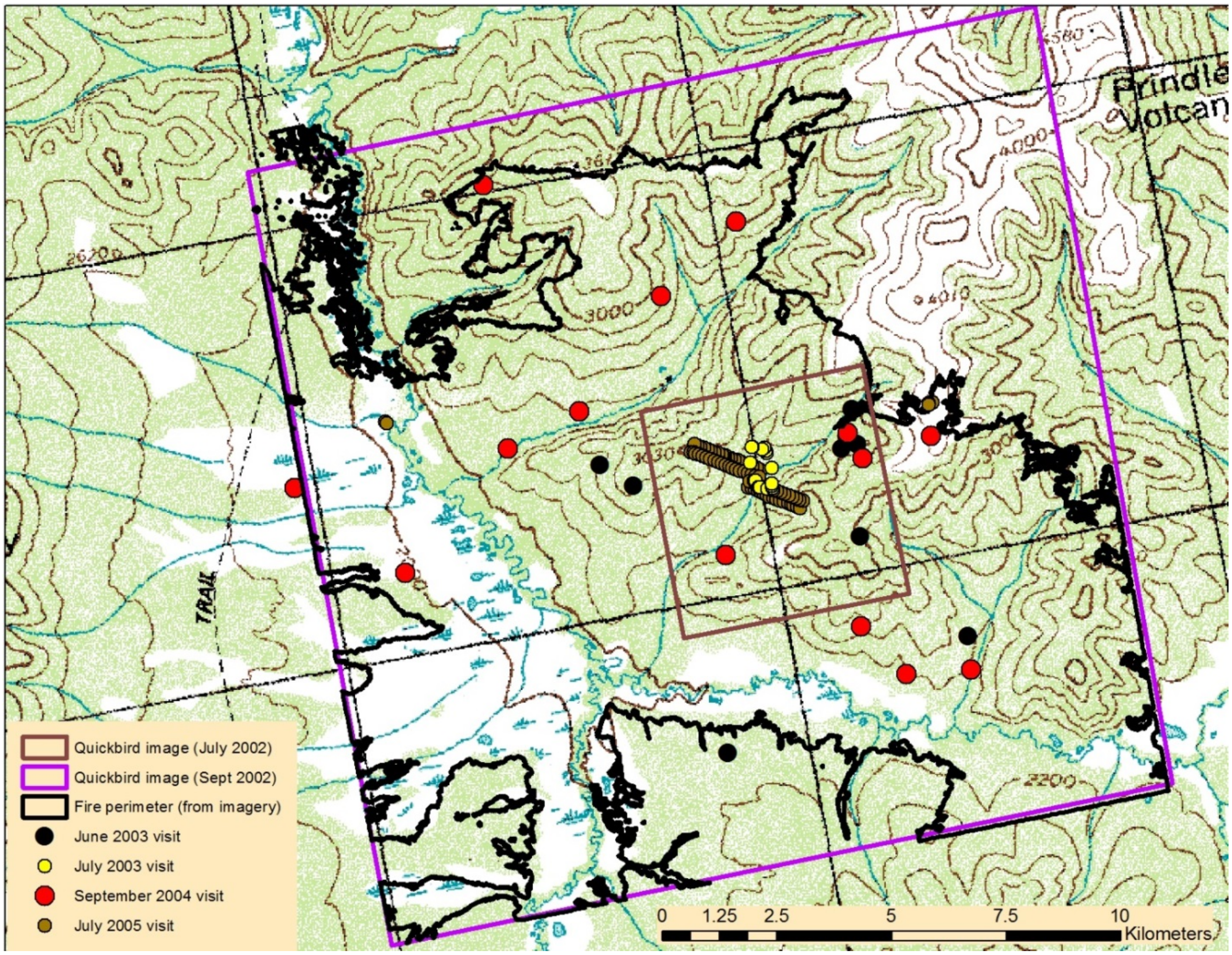


Figure 3. Location of sites visited by helicopter during summers 2003, 2004, and 2005 to ascertain area burned and vegetation type in the Quickbird satellite imagery covering the 1998 East Fork prescribed fire, eastern Interior Alaska.

Methods

Pre-burn vegetation type classes for both of the 1998 burn perimeters (Fig. 2) were characterized from 1:63,360-scale color-infrared (CIR) aerial photos taken on August 1981 and July 1983 (U.S. Geological Survey, EROS Data Center, Sioux Falls, South Dakota). Post-fire type classes were inferred from satellite imagery and CIR photos. On 30 July 2002, we acquired aerial photos covering both burn perimeters (Fig. 2) at 1:12,570 scale from a de Havilland Beaver equipped with a Zeiss RMK-A mapping camera (153 mm focal length) using 9.5-inch Kodak Aerochrome III CIR film 1443© with a D filter (535 nm). The plane was flown 112 km/hr (70 mi/hr) at 305 m (1,000 ft) above ground level with the precision assistance of a radar altimeter along flightlines that provided 30% side and 60% end lap for stereoscopic pairs. The camera was typically set at F5.6 and 1/125 second exposure. Aerial photos were subsequently scanned at 14 microns to produce pixels of 0.33 m, and the pre-burn photos were subsequently resampled to 1 m resolution. We obtained a 272 km^2 Quickbird image (2.6 m multispectral, 0.6 m panchromatic; DigitalGlobe™, Longmont, Colorado; presently distributed by Satellite Imaging Corporation, Houston, Texas, http://www.satimagingcorp.com) covering most of the larger burn perimeter (Fig. 2) taken on 15 September 2002; earlier scenes of the entire study area that summer were rejected by the producer because of cloud cover tolerances. We subsequently purchased a ~25 km^2 Quickbird image taken on 14 July 2002 for more direct phenology comparison to aerial photos on a subset of the larger image (Fig. 2). This 25 km^2 imagery footprint became the study area for inference on vegetative change for the entire 1998 burn.

In October 2002 we used a helicopter to obtain 9 global position system (GPS) coordinates with a handheld Trimble GPS (model not recorded) and portable beacon antenna for ground control points (GCP) at large snags that cast shadows and rocks to obtain approximately 2-m precision for the larger September 2002 Quickbird image (Fig 2). We used the GCPs to orthorectify the image for a base map using Geomatica Orthoengine software (PCI Geomatics, Richmond Hill, Ontario, Canada) for a base map that was geometrically corrected to remove terrain distortions. We conducted 3 subsequent field visits by helicopter to obtain GPS coordinates with a Trimble GeoExplorer3 handheld GPS for verifying reflectance signatures of vegetation types, burned ground, and exposed soil at specific sites (Fig. 3): 27 June and 28 July 2003 (included hovering for aerial photos where landing was not feasible and ground visits across the larger image; n = 26), September 2004 (hovering and ground visits across the larger image; n = 28), and 27–28 July 2005 (systematic ground visits along 2 transects in the smaller image; n = 83). At each site we recorded a digital image; noted whether the site burned; and characterized cover type (see below), canopy height, and species composition. In 2003 and 2004 we subjectively located sites within specific vegetation clumps to aid typing of polygons derived from imagery, including some outside the July image but within the larger September image for purposes of typing burned area. In 2005 we collected data every 100 paces along transects, which resulted in several sites on the edge of classified objects (see next paragraph describing object-oriented analysis), limiting their usefulness for type validation. Digital scans of pre-burn aerial photos were orthorectified from the base map for supervised classification of unburned sites in the July 2002 image footprint, whereas the post-burn photo scans were used to verify cover type of ground control points in the supervised classification of the satellite image.

Data storage and software capabilities to handle large files are important considerations as imagery resolution increases. We chose a post-burn photography scale of roughly 1:12,000 because it provided high resolution, but it resulted in large files (900 MB per frame). The file size of the September 2002 Quickbird scene fused with the panchromatic layer was 7.0 GB at 16 bit or 3.5 GB at 8 bit. GIS specialists with DOF developed a vegetative classification with eCognition®, version 3.0 (Definiens® Imaging, Munich, Germany; presently distributed by Trimble® Geospatial Imaging, Sunnyvale, California, http://www.ecognition.com), which is PC-based software that performed object-oriented analysis on image files up to 2GB to produce polygons composed of pixels of similar value (Baatz et al. 2001). This version of the software was limited to 2 million polygons, so to reduce processing time we divided the September 2002 scene into 9 parts for analysis. Objects (polygons) were derived by segmenting fused and multispectral Quickbird imagery (wavelength color bands 1–4) based on spectral homogeneity and shape and then classified on the resulting layer values (i.e., mean band 1) for the object. We used a normalized difference vegetative index (NDVI) to define vegetative response after fire disturbance (Kasischke and French 1995). We added an NDVI layer ([band 4 − band 3] / [band 4 + band 3]) to aid the segmentation process and create objects that corresponded more closely with the shape of the vegetated areas. Object shape is partly user-controlled by level of resolution; we chose a relatively coarse resolution produced by a segmentation level of 40 as a compromise of resolution and computation time.

Vegetated cover types were defined to at least Level II of Viereck et al. (1992) for the pre- and post-burn classifications with typing to Level III where feasible. We initially defined pre-burn types as alpine tundra, grass (commonly *Calamagrostis canadensis* but also including fireweed [*Epilobium* spp.] and other forbs), shrub (mix of low and tall ≤3 m), deciduous forest that was predominantly aspen, closed spruce forest, open spruce forest and woodland including ericaceous, and open spruce forest and woodland. These types were consolidated to alpine tundra, grass, shrub, deciduous forest, and coniferous forest for analysis. Post-burn types were alpine tundra, grass, shrub (low and tall, including deciduous tree seedlings or sprouts), and coniferous forest. We performed a supervised classification of the pre-burn photos through photo interpretation and use of helicopter visits to unburned sites. We performed a supervised classification of the post-burn image by using sites visited in 2005 for training in polygon typing and withheld 12 sites for estimating classification error. Terrain covariates of elevation (m), slope (deg.), and aspect (0 if flat and degrees converted to quadrants: NE = 1–90 deg, SE = 91–180, SW = 181–270, NW = 271–360) were derived from an ASTER digital elevation model (U.S. Geological Survey) of 15 m input resolution resampled and smoothed to 6 m for the 25 km^2 July image. The raw Quickbird imagery for the East Fork burn was archived at the Fairbanks office of DOF (owner of site license) whereas digital photos scans and derived digital products were archived at the Fairbanks office of DWC by the lead author.

We identified 4 issues that limited our ability to assess vegetation type change and the effects of terrain covariates on the response variable (post-fire vegetation). First, the sample size of ground truthing points was relatively small and of compromised quality (described above). We retained a small number of points for estimating classification error in the post-burn image (Table 1) and none for the pre-burn image. Second, we did not have an estimate of fire severity, which is typically higher in conifer forest than in deciduous forest (Epting 2004:91). High severity fire tends to favor deciduous forest establishment over conifer (Epting 2004, Johnstone and Kasischke 2005). Methods exist to use pre-burn and immediate post-burn imagery to calculate a differenced Normalized Burn Ratio (dNBR; Key and Benson 2006) with appropriate cautions at high latitudes (Verbyla et al. 2008). Several researchers recommend validation of dNBR values with observed severity classes prior to inferring ecological effects of fire in the boreal forest of Alaska (Hoy et al. 2008, Kasischke et al. 2008, Murphy et al. 2008). We could not locate immediate post-burn imagery (fall 1998 or summer 1999) suitable for burn severity assessment. Further, our assessment began 4 years post-fire, which would have also hindered direct evaluation of fire severity (mineral soil exposure) during site visits because of early revegetation by grass. Third, use of eCognition resulted in unique polygons for the pre- and post-burn images, which prevented direct change detection. This forced our analysis back to the 1-m pixel level, which greatly increased computational complexity. Finally, we did not have validated spatial data on soil characteristics (organics, pH, etc.), which can influence fire severity and thus vegetation species presence and growth rate (Hollingsworth et al. 2013). The combination of these 4 issues led us to take a measured approach toward analysis and use caution when interpreting results.

Table 1. Transition matrix[a] for change in classified vegetation type from before to after prescribed fire in a 25 km^2 portion of the 1998 East Fork burn, Interior Alaska. Pixels lacking surface data (under cloud cover) and nonvegetated pixels were removed prior to analysis.

Pre-burn classification	Post-burn classification: Conifer	Deciduous	Alpine tundra	Grass	Shrub	Total pixels	p_i
Conifer	**0.064**	**0**	**0**	**0.407**	**0.529**	18,170,056	**0.797**
Deciduous	**0**	**0**	**0**	**0.294**	**0.706**	412,423	**0.018**
Alpine tundra	**0**	**0**	**1.0**	**0**	**0**	470,931	**0.021**
Grass	**0.171**	**0**	**0**	**0.263**	**0.566**	329,124	**0.014**
Shrub	**0.092**	**0**	**0**	**0.338**	**0.570**	3,415,295	**0.150**
Total pixels	1,531,629	0	470,931	8,751,236	12,044,033	22,797,829	
p_f	**0.067**	**0.000**	**0.021**	**0.384**	**0.528**		**1.000**

[a] Proportions (p_i = pre-burn, p_f = post-burn) are shown in bold. For example, marginal totals indicate that coniferous forest composed 79.7% of the image before the fire and shrub composed 52.8% of the image after the fire, whereas the proportion of pixels classified as conifer before the burn being classified as grass after the burn was 0.407.

We began our analysis by developing a transition matrix to explore the changes in vegetation from pre- to post-burn. The July 2002 image was composed of 23,167,790 1 m^2 pixels with 1.6% having cloud cover, leaving 22,797,829 pixels with useful data. A transition matrix was produced with pixel counts extracted from classified objects to determine the proportional changes in cover types before and after the burn. This analysis did not consider classification error. Our central interest was the transition of a burned conifer dominated landscape to willow shrubs and hardwood samplings for moose browse, with a secondary interest in transition of burned conifer to grass-herbaceous cover for small mammals (forage base for avian and mammalian predators). We focused our modeling effort on ascertaining what variables may affect whether conifers transition to shrub or grass when burned, with grass potentially converting to willow shrubs and hardwood seedlings when woody seeds sprout or grow beyond competition with grass. In the pre-burn landscape, 18,170,056 pixels were classified as coniferous forest. We found 9,456 of these data had aberrant covariate values (−999) for elevation and slope along latitude and longitude lines near the image border. These pixels were eliminated because they showed no patterns relative to the covariate values and made up a minor portion (0.05%) of the study area. Of these 18,160,600 data, 1,157,422 (6.4%) were conifer pixels that had not burned and remained conifer. Therefore, the sample size for modeling was the remaining 17,003,178 pixels that changed from conifer to shrub or grass. We assume terrain influences whether or not vegetation is present, the type of vegetation if present, and the growth rate of vegetation (Chapin et al. 2006a): elevation defines growing season length (shorter with increase in elevation); aspect influences soil warmth and photosynthetic potential (southern component favorable); and slope influence drainage (typically drier on steeper sites, although soil type can influence water retention capacity). To facilitate the use the circular variable, aspect, as a covariate we converted it into a categorical variable: N2E (0–89°), E2S (90–179°), S2W (180–269°), W2N (270–359°), and zero if slope = 0.

We used generalized linear models with a binary response (i.e., grass or shrub) and a logit link function (essentially a maximum likelihood version of the classic logistic regression model) to model the probability of conifers changing to shrub or grass. We analyzed these models using the GLIMMIX procedure in SAS Version 9.2 (SAS Institute, Inc., Cary, North Carolina, http://www.sas.com/en_us/home.html). The most complicated (global) model in our candidate set included the 3 main effects of elevation, slope, and aspect and the interaction between slope and elevation. We performed model selection using Akaike's information criterion (AIC; Akaike 1973). Subsequent to selecting our candidate set of models, we prepared histograms and boxplots using the Univariate and Boxplot procedures in SAS 9.2 to explore the distribution of terrain covariate values and their relationship to the response variable (conifer to shrub or grass).

Our initial models assumed no spatial correlation among the responses (nonspatial), and we subsequently developed models that accounted for spatial correlation. The spatial models are more appropriate, but we present both to illustrate the effect of accounting for the spatial correlation among data and because we used nonspatial model results to focus our spatial analysis. Modeling spatial correlation in the response variable is important because failing to do so leads to negatively biased variances and *P*-values and inflated Type I error (i.e., covariates may appear to be significant when they are not).

Modeling spatial correlations for large data sets presents computational problems because of vast requirements for computer memory when inverting large covariance matrices. Using a variogram model (e.g., exponential) to account for the spatial correlation, we were only able to handle simple random samples of about 5,000 pixels. We decided to use the low rank radial smoother option, GLIMMIX, TYPE=RSMOOTH ( ), to model the spatial correlation because this approach is computational efficient (Ruppert et al. 2003, Schabenberger and Gotway 2005). Because of convergence problems associated with random effect variances near zero, we needed to rescale the spatial coordinates (Schabenberger and Gotway 2005). The NOLOG suboption available in SAS 9.2 was used to eliminate a dependency of the RSMOOTH approach on coordinate scaling. To guard against misinterpreting model results, in particular for inference beyond the study area, we divided the study area into 2 sections of fairly similar size and landscape along an east-west line, analyzed them independently, and compared results. This approach also effectively halved the size of the data set when analyzing spatial models reducing computational complexity. To aid in interpreting model results we generated frequency counts for covariates within each section using Proc Univariate.

Results

We used 2002 aerial photos to verify accuracy of the burn perimeter derived from the NDVI signature of the imagery. The area inside the revised burn perimeter (13,794 ha) within the 272 km^2 September 2002 image (Fig. 2) was 89% burned area, 10% unburned, 1% water (streams), and <0.1% unknown (Appendix). After 4 growing seasons willows had grown to 2 m and aspen to ≥3 m by sprouting from existing root systems, but post-fire sprouting of woody plants from seed was not readily apparent at sites we visited. Using our specified segmentation level and the reduced suite of classes, eCognition created 277,962 objects for pre-burn images from aerial photography (mean 93 m^2, range 1–593,553 m^2) and 251,417 objects (mean 92 m^2, range 0.01–962,290 m^2) for the post-burn satellite image (actual area 23,165,559.8 m^2). The post-burn classification of the 25 km^2 July 2002 image produced more small objects and fewer large objects than the pre-burn classification (Fig. 4), including 9,911 objects <1m^2 (4,301 m^2 total = 0.02% of classified area) and 4,131 objects <0.36 m^2 (resolution of 60 cm panchromatic; 134.5 m^2 = 0.001%; Fig. 5). These minute objects resulted from the occasional spatial mismatch between pre- and post-burn objects created from images of different resolution (Fig. 6).

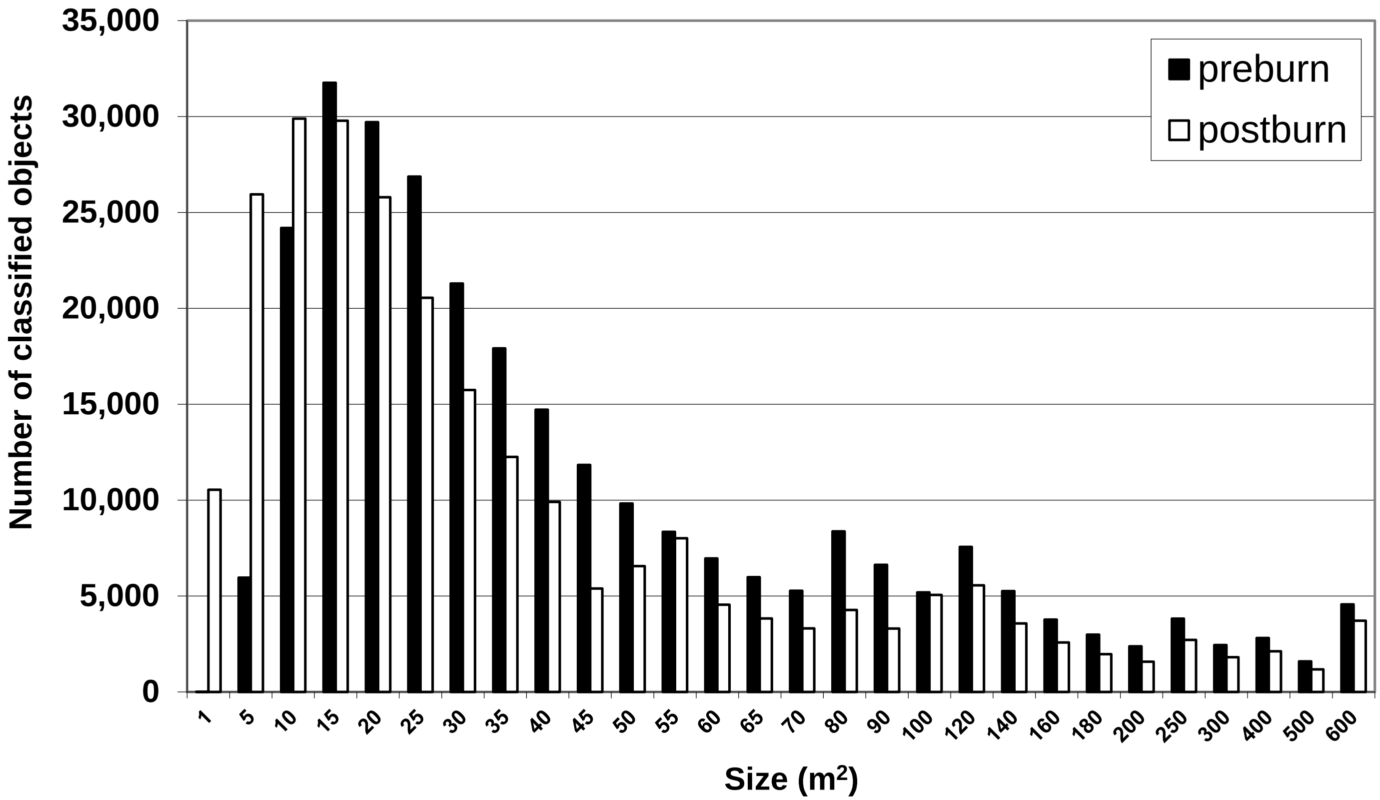


Figure 4. Size distribution of objects (polygons comprised of pixels) created by eCognition software in supervised classifications of pre-burn (1 m resolution) and post-burn imagery (0.6 m resolution) for a 25 km^2 portion of the 1998 East Fork prescribed burn. Bin intervals are variable to illustrate detail. Category 600 is all objects >600 m^2.

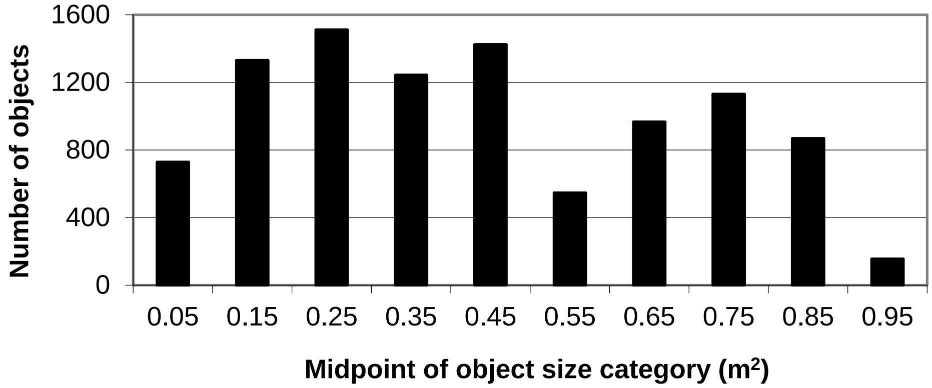


Figure 5. Size distribution of classified objects (polygons comprised of pixels) <1 m^2 created by eCognition software in supervised post-burn classifications (0.6 m resolution) for a 25 km^2 portion of the 1998 East Fork prescribed burn.

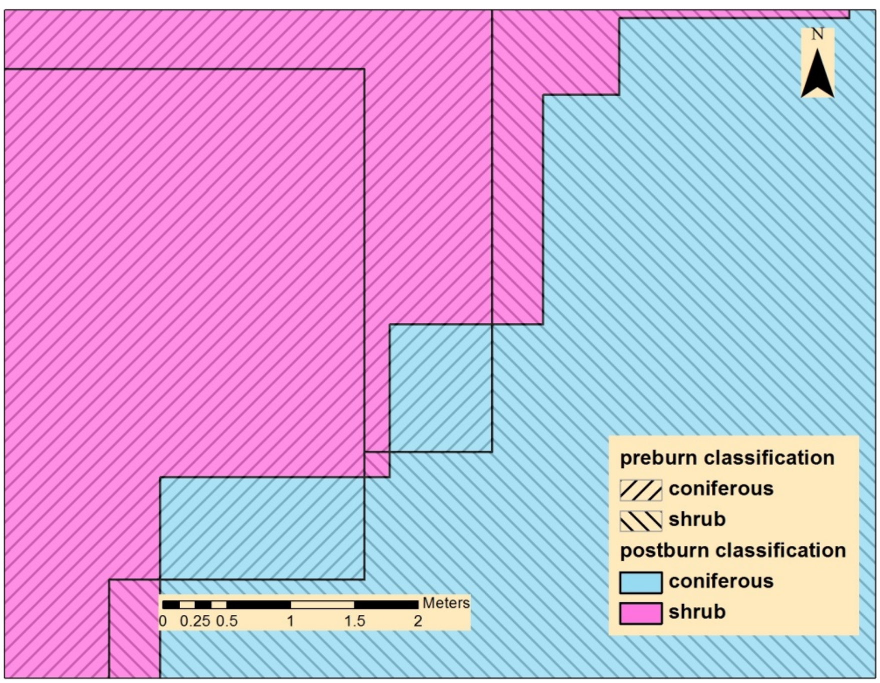


Figure 6. Example of occasional spatial mismatch between pre-burn objects classified from scanned aerial photos (1 m resolution) and post-burn objects classified from Quickbird imagery (0.6 m panchromatic resolution).

The transition matrix of pre- and post-burn types revealed that 6.4% of pixels were defined as conifer forest in both classifications (Table 1, Fig. 7) for a minimum estimate of unburned spruce inclusions. This is a minimum estimate of unburned because grass or shrub could have regenerated in 4 growing seasons and approximates the estimate of 10% unburned from the NDVI analysis. The pre-burn classification of 80% conifer, 15% shrub, and 1% grass changed to

7% conifer, 53% shrub, and 38% grass (Table 1). Given the primary intent of the burn to produce seral conversion of conifer to herbaceous species or shrubs, we noted that 41% of conifer pixels transitioned to grass and 53% transitioned to shrub, which included young aspen. Regardless of whether a pixel actually burned, 26% of grass pixels before the burn remained grass and 57% of shrub remained shrub. We found that alpine remained alpine, nothing converted to alpine, the few (1.8%) deciduous forest pixels became grass (29%) or shrub (71%), and nothing became deciduous forest (Table 1). We urge caution in interpreting minor classes because of limited data for estimating error in classification (Table 2).

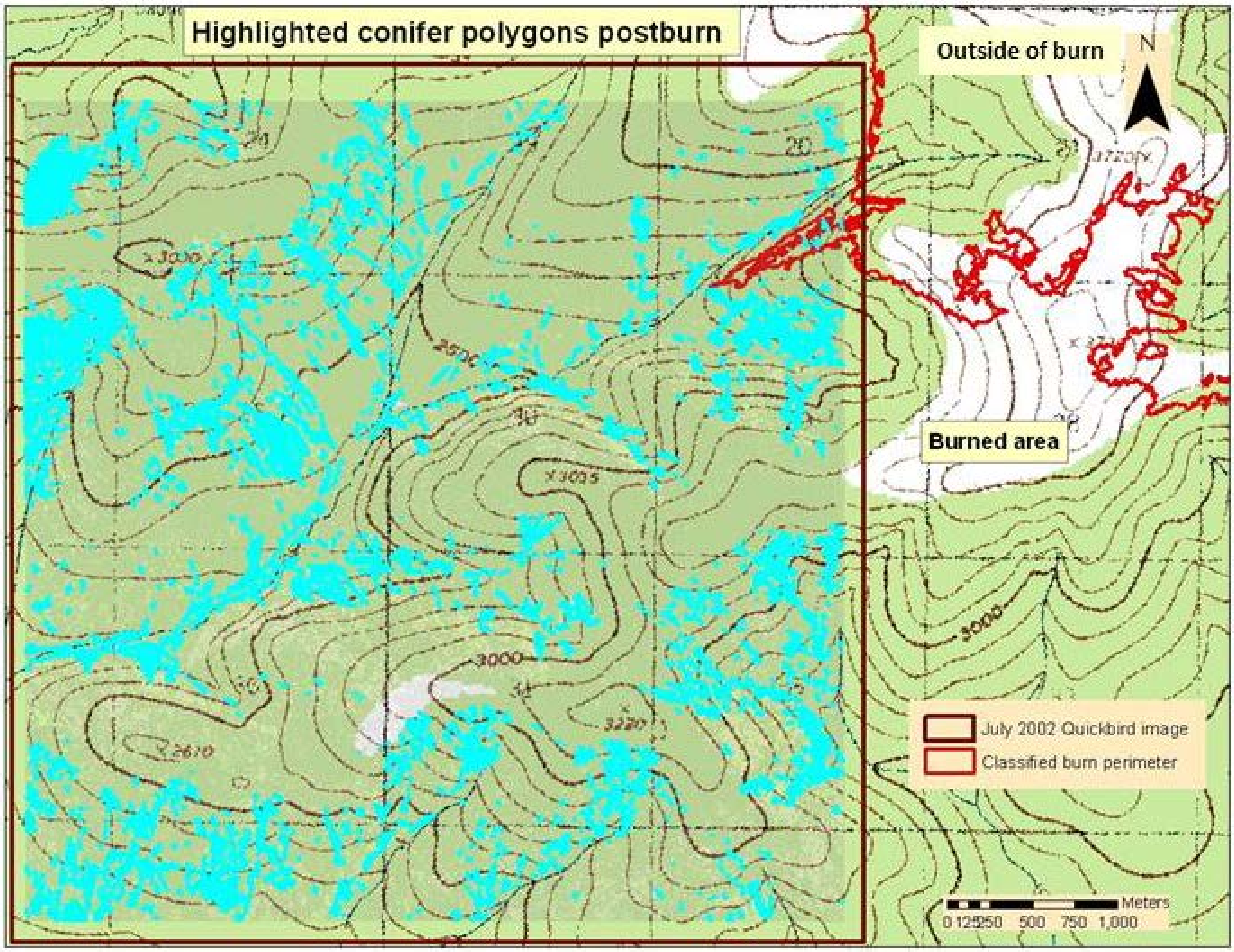


Figure 7. Highlighted pixels from classified objects for which conifer cover pre-fire remained as conifer cover following the 1998 East Fork prescribed burn in eastern Interior Alaska. The northeast corner of this figure is outside the burned area.

Table 2. Error matrix of correspondence counts and resulting probability of accuracy of a supervised classification compared with the field validation at the same site for a 25 km^2 July 2002 post-burn satellite imagery within the 1998 East Fork prescribed burn, eastern Interior Alaska. These were a random subset of sites visited that were not used for the supervised post-burn classification.

	Field validation			
Classification	Conifer	Shrub	Grass	User accuracy
Conifer	0	1	0	0
Shrub	1	6	0	0.86
Grass	0	0	4	1.0
Producer accuracy	0	0.86	1.0	0.83[a]

[a] Overall classification accuracy.

The relationship between the terrain covariates associated with pixels that converted from conifer to grass and those converting from conifer to shrub exhibited substantial consistency between and within the northern and southern sections of the study area (Table 3). However, some differences are notable. Elevation had a symmetric distribution in the northern section, whereas it showed a slight skew to lower elevations in the southern section (Fig. 8a) and the mean elevation for the northern section was about 100 m higher than for southern section (Table 3). The mean elevation and associated percentiles for pixel converting from conifer to shrub were slightly smaller than those for conifer to grass. The maximum slope in the northern section is 37° whereas the maximum in the southern section is 82°. Although the mean slope is quite similar between sections, the southern section has a relatively small proportion of steep slopes barely discernable on histograms (Fig. 8b). In the northern section there is a slight tendency for the conifer to shrub transition to be associated with steeper slopes (Table 3). In the southern section the steepest slopes are associated with the transition to shrub; however, slopes of ~25–60 degrees are dominated by the transition to grass. Grass areas in the northern and southern section had nearly identical aspect, which is predominantly to the SSW with 50% of the slopes facing SSE to W (Fig. 8c). The mean aspect for pixels converting to shrubs in the southern section is similar to those converting to grass; whereas, those converting to shrubs in the northern section tend to have a slightly more southerly orientation (Table 3).

Table 3. Summary statistics for the distribution of terrain covariate values associated with pixels that converted from conifer to grass or shrub in northern and southern halves of the 25 km^2 study area following a 1998 prescribed burn in eastern Interior Alaska.

	Elevation (m)				Slope (degree)				Aspect (degree)			
	Northern		Southern		Northern		Southern		Northern		Southern	
Variable	Grass	Shrub	Grass	Shrub	Grass	Shrub	Grass	Shrub	Grass	Shrub	Grass	Shrub
Mean	891	872	783	754	11	12	9	10	200	185	204	200
Median	895	866	776	750	12	12	9	9	207	180	207	198
25th percentile	852	828	736	705	8	9	6	6	153	135	153	157
75th percentile	930	912	830	798	13	15	12	13	270	255	270	248

(a)

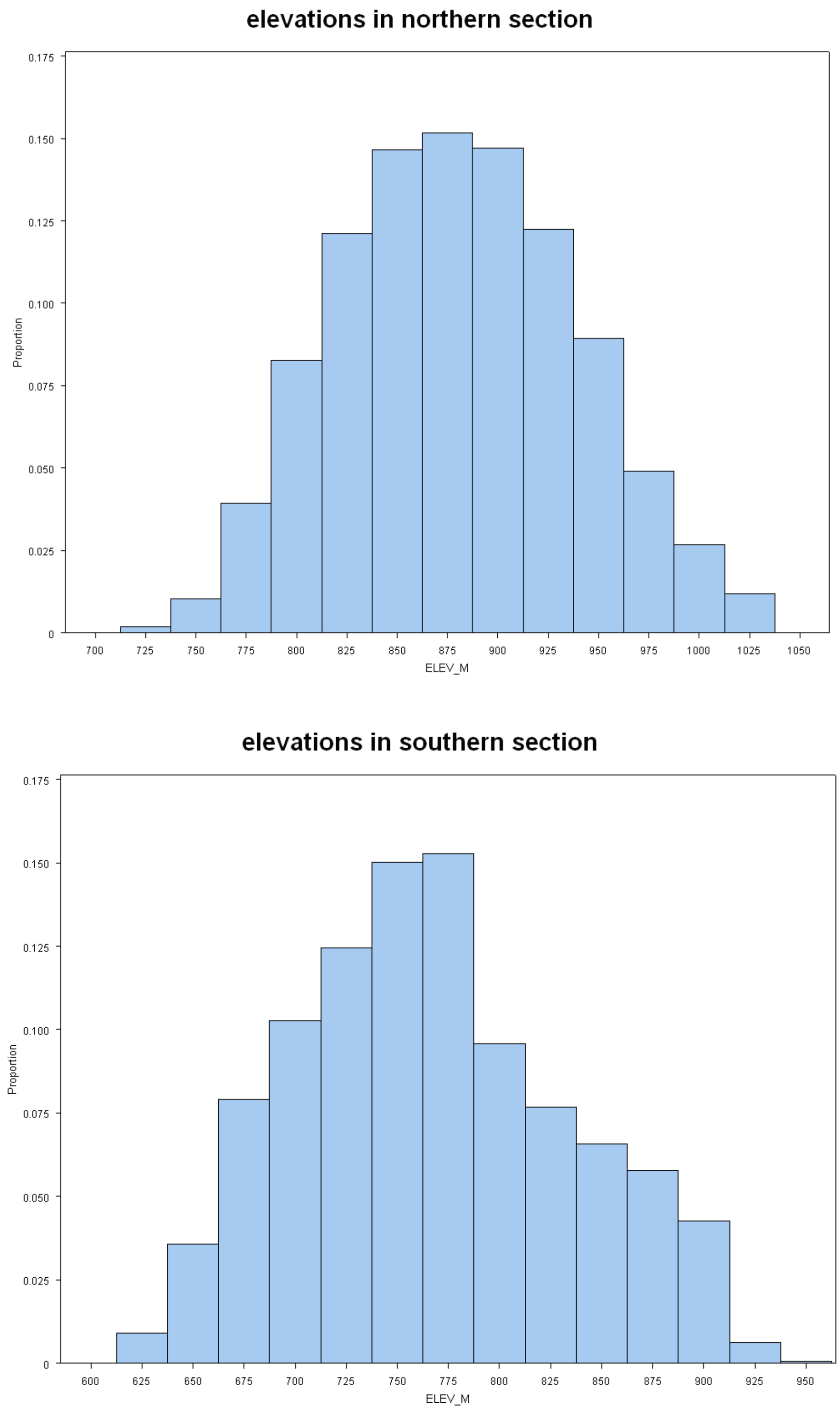
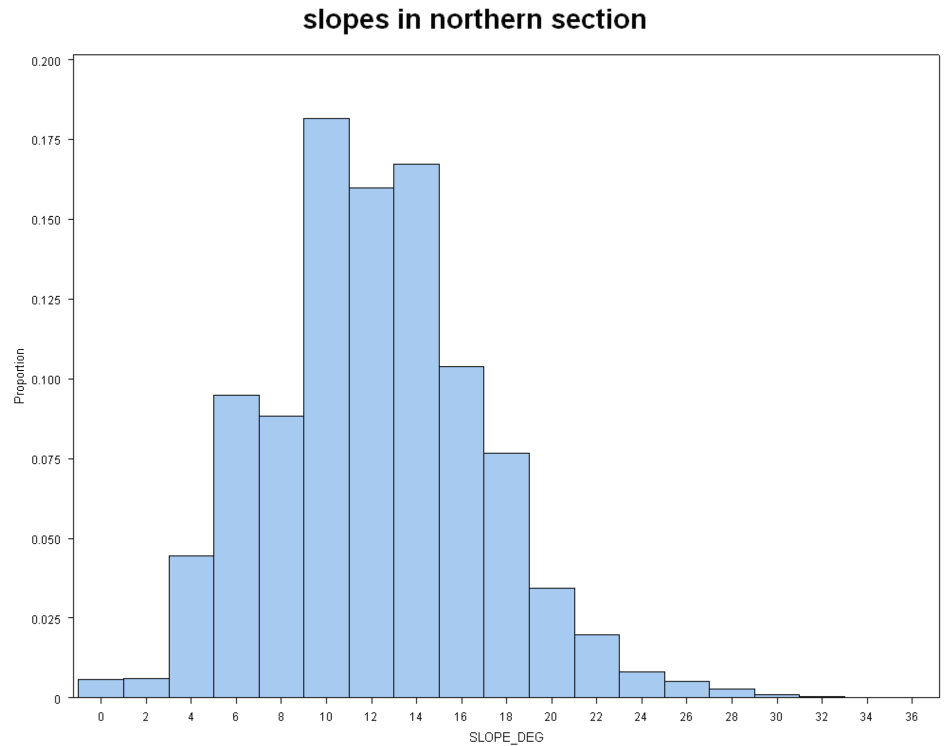
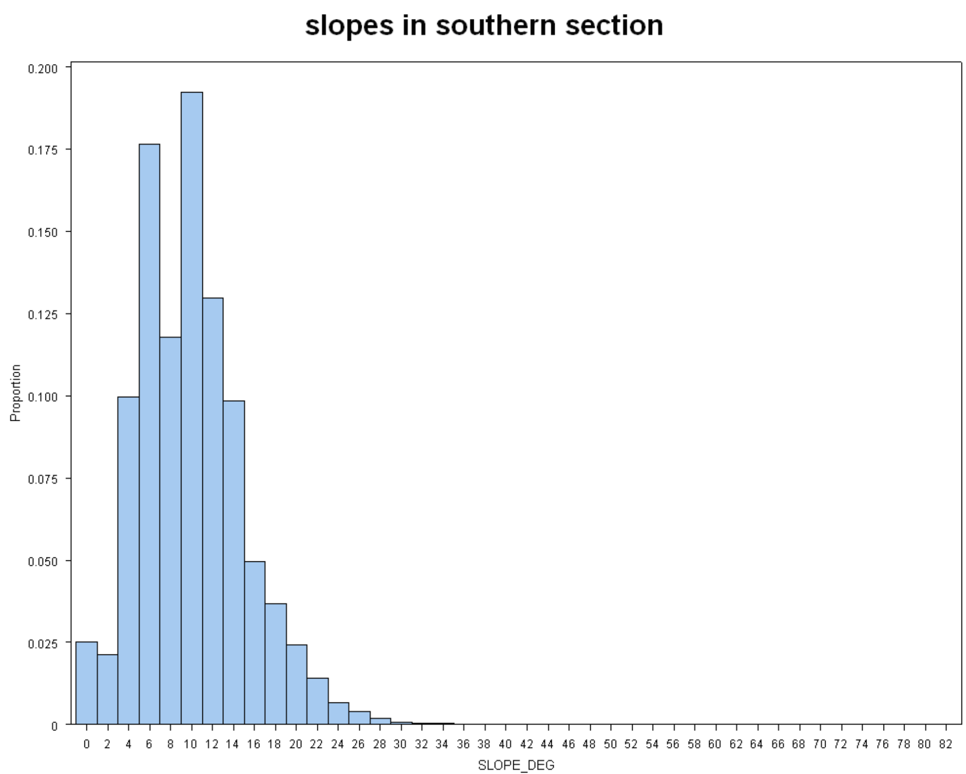


Figure 8. Histograms of elevation (a), slope (b), and aspect (c) between northern and southern halves of the 25 km^2 study area following a 1998 prescribed burn in eastern Interior Alaska. Extent of x-axes indicates presence of data.

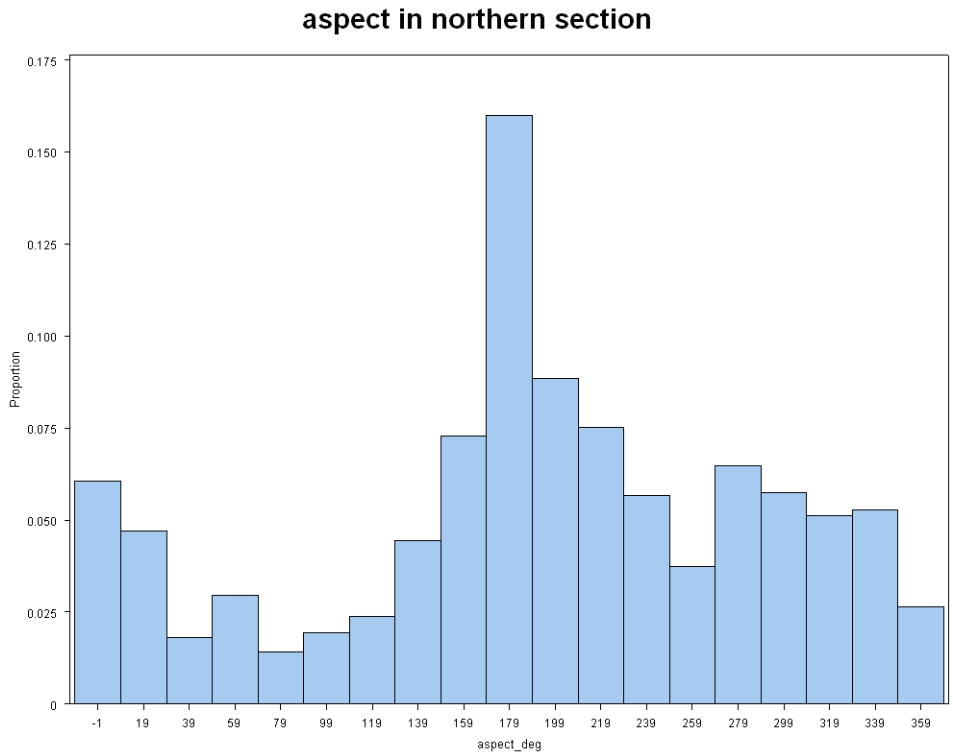
(b)

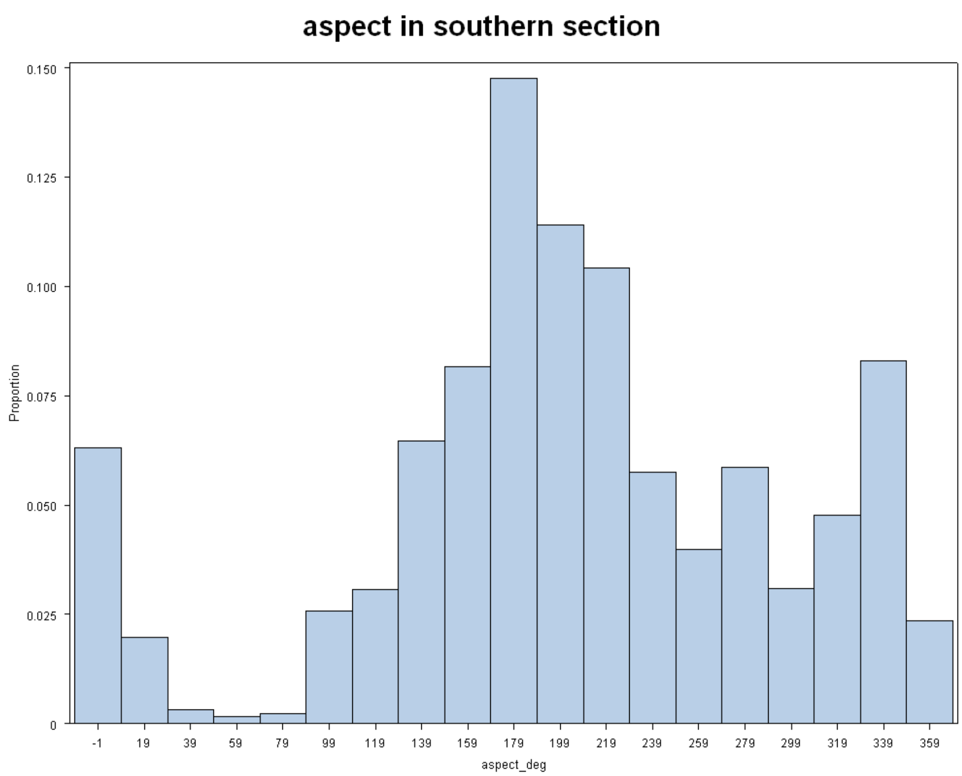




(continued) **Figure 8.**

(c)





(continued) **Figure 8.**

The most complicated nonspatial model we considered (global model of 3 main effects and the interaction between elevation and slope) had the lowest AIC score (Table 4). However, aspect yielded inconsistent results between sections, with west facing slopes having lower probability of changing to shrub in the northern section and north facing slopes having lower probability of changing to shrub in the southern section. In addition, the sign of the coefficients for each category of aspect were inconsistent between the northern and southern sections. Finally, compared to the effects of elevation and slope, the effect of aspect on the transition probabilities was relatively minor (5–10 percentage points). As a result, we considered these results unreliable or of limited predictive capability so omitted aspect from the model.

Table 4. Akaike's information criterion (AIC) scores for nonspatial models. Lowest AIC scores indicate the most parsimonious fit in each section.

Model covariates	Northern section	Southern section
Elevation+slope+aspect+elevation:slope	11193484	10943353
Elevation+slope+aspect	11259354	11075931
Elevation+slope+elevation:slope	11292233	11009020
Elevation + slope	11368557	11139962

The nonspatial model we considered most reliable is that with elevation, slope, and their interaction, which was the next most favored model in the southern section (Table 4). We believe an interaction between elevation and slope may produce an ecological condition unique of that produced from either individual parameter, justified its use as a modeling factor. The parameter estimates for elevation, slope, and their interaction were reasonably consistent between sections (Table 5). Because we did not model the spatial correlation in this instance, we did not expect the standard errors and *P*-values for this model to be reliable. We reported them in Table 5 to compare with the preferred spatial model. From this nonspatial model we estimated the probability of conifer changing to shrub as

$$p_{conifer \rightarrow shrub} = \frac{e^{\beta_o + \beta_1 \times elev + \beta_2 \times slope + \beta_3 \times elev \times slope}}{1 + e^{\beta_o + \beta_1 \times elev + \beta_2 \times slope + \beta_3 \times elev \times slope}} \qquad \text{Equation 1}$$

and the probability of conifer changing to grass as

$$p_{conifer \rightarrow grass} = 1 - p_{conifer \rightarrow shrub} \qquad \text{Equation 2}$$

Table 5. Parameter estimates for the nonspatial model of elevation, slope, and their interaction as an effect of vegetation response after fire.

Parameter	Section	Estimate	SE	*P*-value
Intercept (β_0)	Northern	−12.214	0.029	<0.0001
	Southern	−11.194	0.01936	<0.0001
Elevation Coef (β_1)	Northern	0.014	0.00003	<0.0001
	Southern	0.015	0.00003	<0.0001
Slope Coef (β_2)	Northern	0.610	0.002	<0.0001
	Southern	0.619	0.0019	<0.0001
Elev:Slope Coef (β_3)	Northern	−0.0008	−0.0009	<0.0001
	Southern	2.808E-6	2.423E-6	<0.0001

The magnitude of the effects was relatively large in the nonspatial model (Fig. 9). When interpreting model results, the range of elevations and slopes that represents 95% of the pixels in each analysis section (Table 6) should be taken into account because model results in regions outside these bounds correspond to only small portions of the study area and are not as well supported by the data (Figs. 8 and 9). For example, the potential for conversion from conifer to shrub for the majority (95%) of the combinations of elevation and slope observed in each study area sections generally increases as elevation decreases and as slope increases in both sections of the study area (inside white box, Fig. 9). This general trend is complicated by the interaction of elevation and slope leading to some exceptions for these elevation and slope combinations and different trends for the rarer 5% of combinations. For example, the predicted probability of going from a conifer to a shrub for these rarer combinations was greatest at the highest elevations and steepest slopes as well as at the lowest elevations and shallowest slopes (Fig. 9: darkest areas outside the white box outline).

(a)

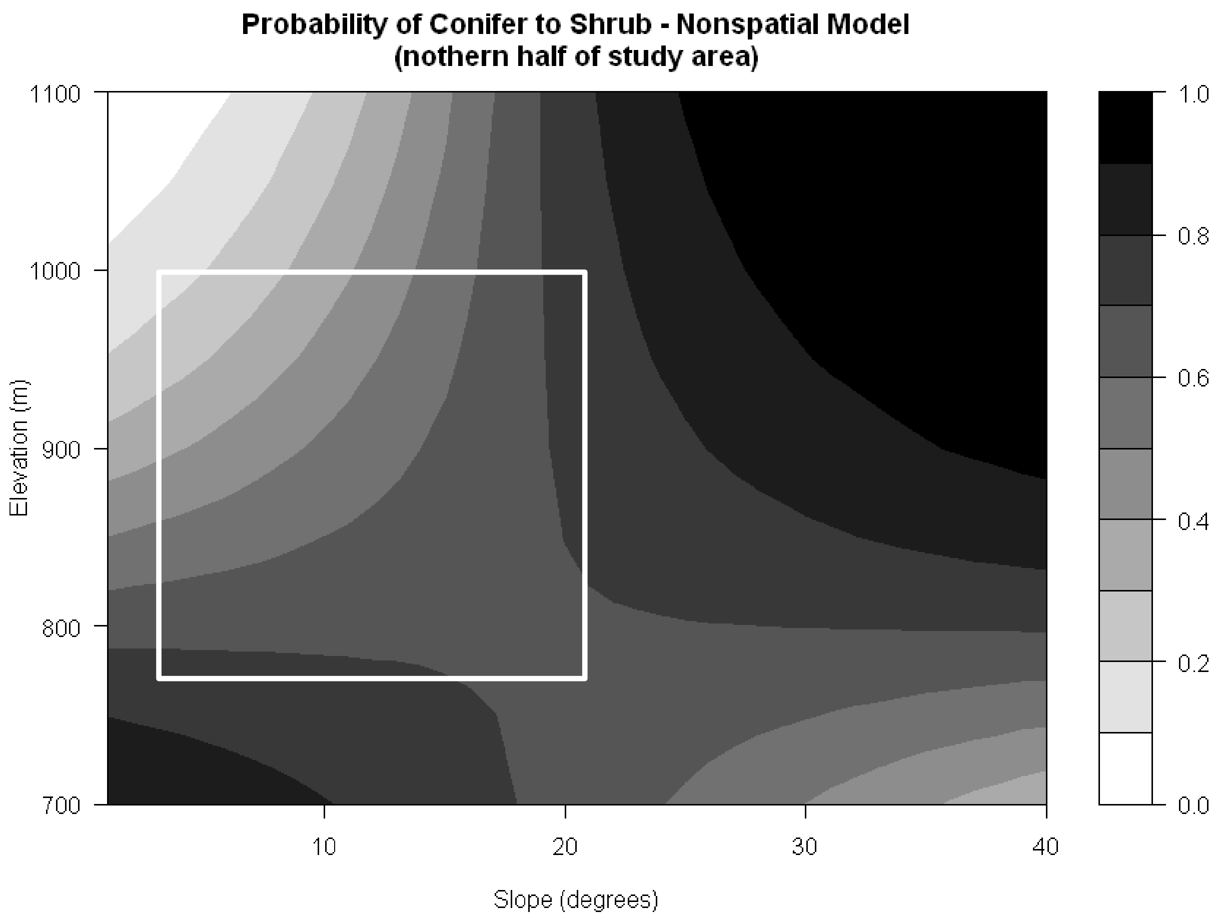
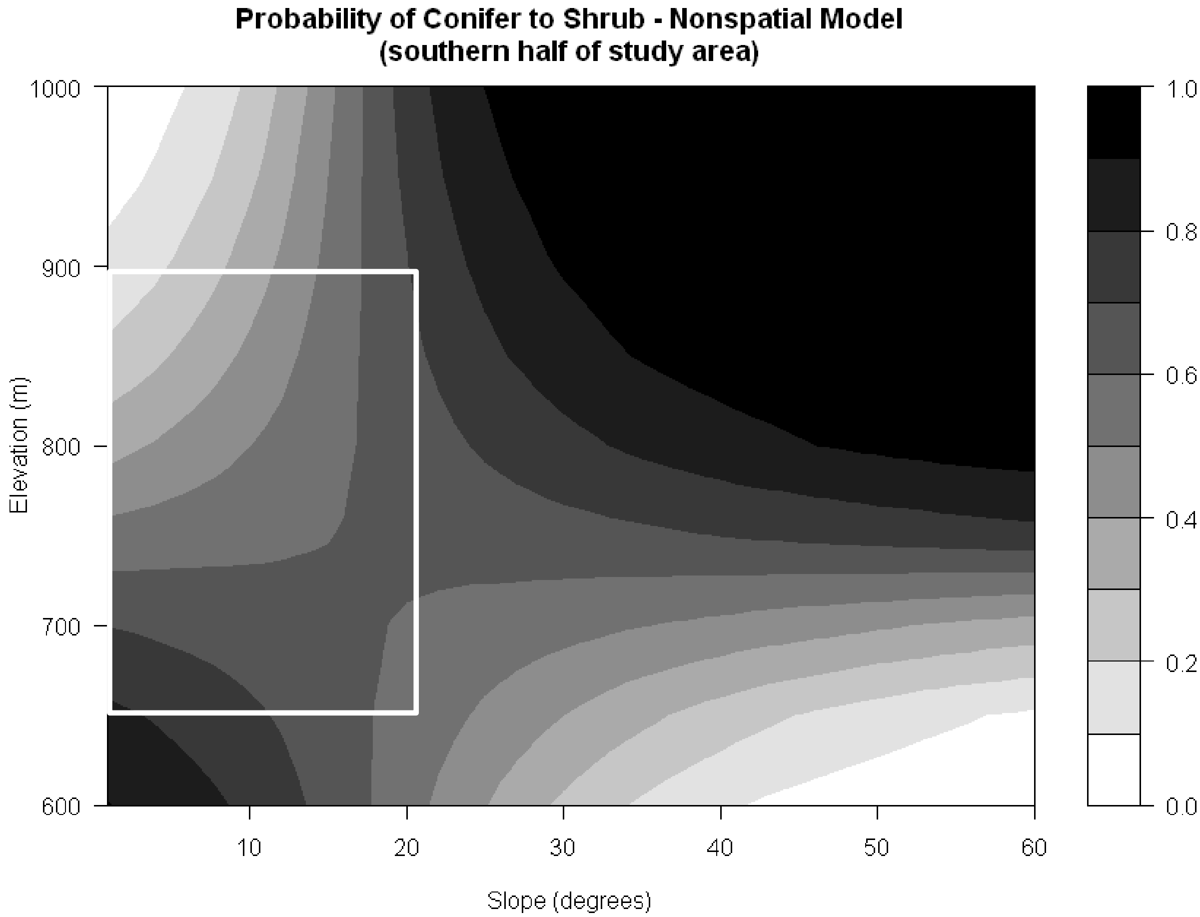


Figure 9. Contour plot of the probability in the nonspatial model of conifer changing to shrub as a function of elevation and slope (Table 5) for the northern (a) and southern (b) sections of the 25 km^2 study area following a 1998 prescribed burn in eastern Interior Alaska. The region outlined in white corresponds to combinations of slope and elevation for 95% of the pixels in the study area (Table 6).

(b)



(continued) **Figure 9.**

Table 6. Ranges of elevation and slope that describe 95% of each analysis section for the nonspatial and spatial models.

Section	Northern		Southern	
Parameter	Elevation (m)	Slope (deg)	Elevation (m)	Slope (deg)
Central 95% of data	774–998	3–21	653–897	0–21

We then developed a spatial version of this model using the low-rank radial smoother. We were able to model the spatial correlation in simple random samples of up to 8.3 and 8.2 million data, just shy of the ~8.6 and ~8.4 million pixels in the northern and southern sections, respectively. Parameter estimates were virtually identical for samples exceeding 5 million data, indicating that our model results apply to the sections in their entirety. The main effects of elevation and slope and the interaction of elevation and slope are highly significant in the spatial model (Table 7). We again used Equation 1 to estimate the probability of conifer changing to shrub for each pixel. As with the nonspatial model, the magnitude of effects was relatively large, particularly for elevation (Fig. 10). These estimates are values typical for pixels with the same covariate pattern (i.e., pixel specific random effects were not applied). Both nonspatial and spatial models had significant effects due to elevation, slope and their interaction. The effect sizes for the nonspatial and spatial models were similar for the southern section. The slope effect and the interaction term were not as strong in the spatial model for the northern section. However, the general pattern for the spatial model is similar between sections within the ranges corresponding to 95% of the data (within bounds of Table 7), indicating that conifers are more likely to transition to shrubs than grass at lower elevations with steeper slopes.

Table 7. Parameter estimates for the spatial model of conifer transition to shrub as a function of elevation, slope and their interaction using a low-rank radial smoother.

Parameter	Section	Estimate	SE	*P*-value
Intercept (β_0)	Northern	−13.2826	0.6689	<0.0001
	Southern	−14.5620	0.4135	<0.0001
Elevation Coef (β_1)	Northern	0.01496	0.000034	<0.0001
	Southern	0.01974	0.000029	<0.0001
Slope Coef (β_2)	Northern	0.2917	0.002584	<0.0001
	Southern	0.6968	0.001920	<0.0001
Elev:Slope Coef (β_3)	Northern	−0.00040	2.973E-6	<0.0001
	Southern	−0.00097	2.522E-6	<0.0001

(a)

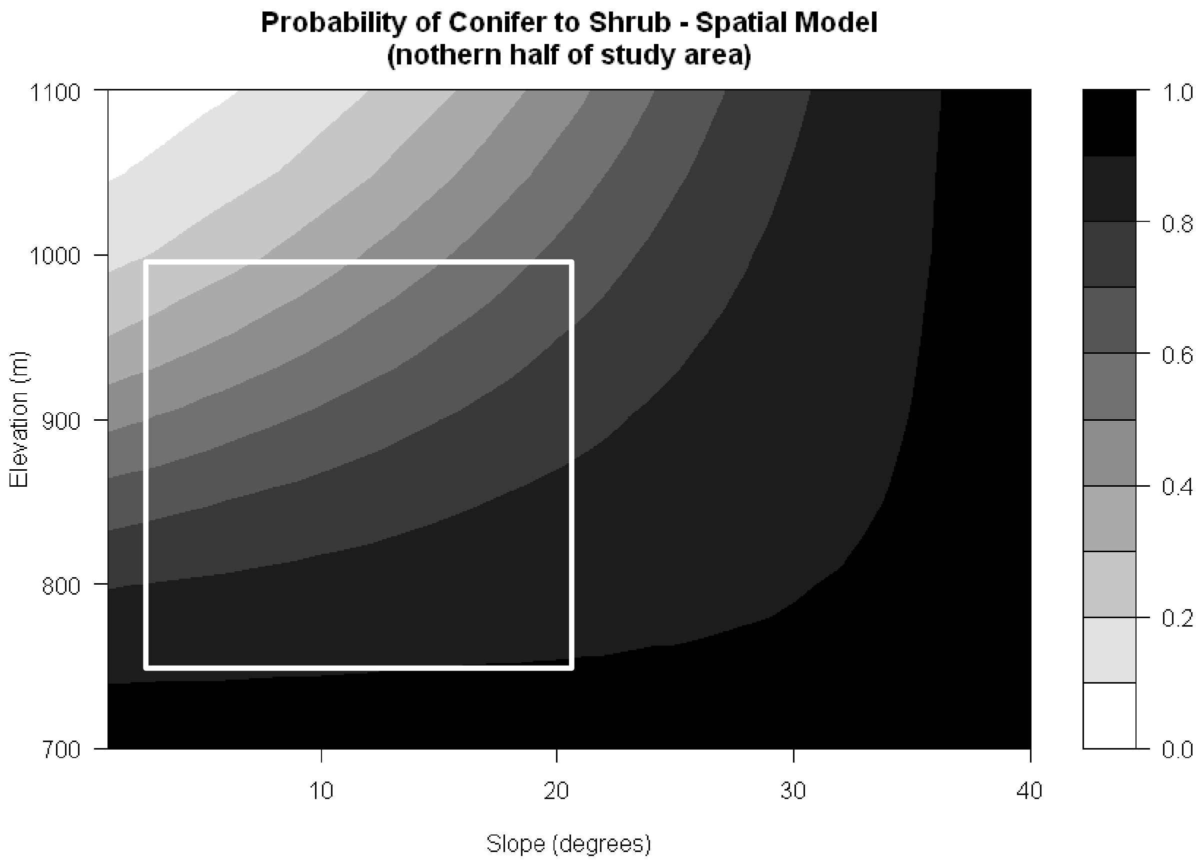
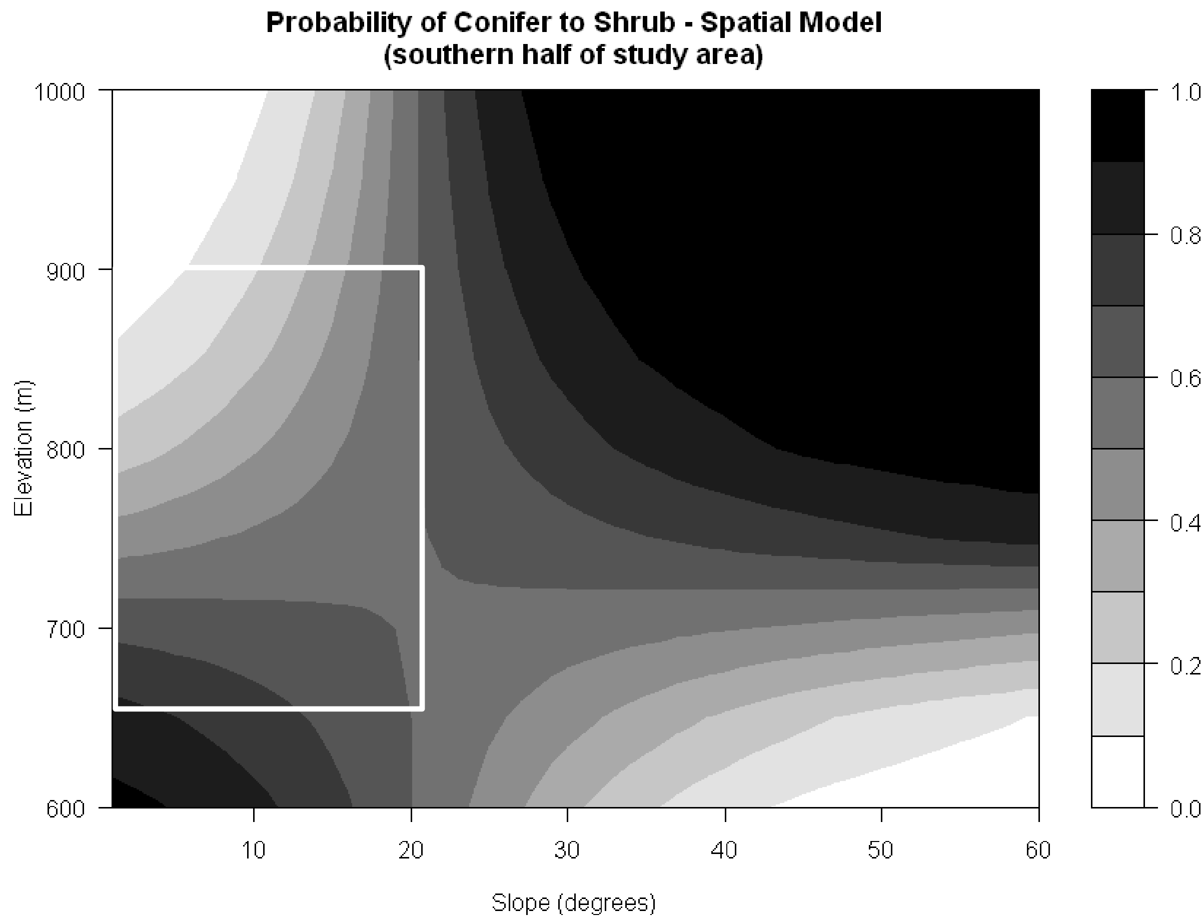


Figure 10. Contour plot of the probability in the spatial model of conifer changing to shrub as a function of elevation and slope (Table 7) for the northern (a) and southern (b) sections of the 25 km^2 study area following a 1998 prescribed burn in eastern Interior Alaska. The region outlined in white corresponds to combinations of slope and elevation for 95% of the pixels in the study area (Table 6).

(b)



(continued) **Figure 10.**

Discussion

Our analysis of the effects of aerial ignition applied over a range of terrain features in the East Fork prescribed burn suggests that elevation and slope, but not aspect, may influence seral conversion of spruce to shrubs and young hardwoods. However, our findings should be applied to planning future burn prescriptions with caution because these data evaluate only one study area. Further, we were unable to include important covariates such as soil characteristics and fire severity in our analysis that may interact with terrain metrics. For example, Epting (2004:92) found a relationship between elevation and fire severity, with lower severity at higher elevations attributed to lower fuel availability in proximity to treeline. He attributed the lack of aspect effect on fire severity to the fact that coniferous forest occurs on both north and south slopes in Interior Alaska, although white spruce may dominate on south-facing aspects and the more flammable black spruce on north-facing or colder sites. Our lack of spatial data on fire severity also precluded an analysis of the relationship between terrain variables and the degrees of fire severity. Understanding the resultant fire severity would provide a mechanistic understanding of vegetation change as a function of root sprouting or seedling establishment by browse species on burned sites (Johnstone and Kasischke 2005) and eventually the relationship between fire severity and browse production and its removal by moose (Lord 2008). Given the model limitations stated above, we hesitate to recommend terrain variables for planning which areas are most conducive to conversion from conifer to shrub.

A second consideration is that the observed relationship with elevation and slope and the seral conversion from conifer to browse species was not simple, which may limit the usefulness of terrain covariates in our study for planning burn areas. Finally, our initial evaluation after 4 growing seasons was primarily on sprouts. Johnstone et al. (2004) found that the majority of tree seedling establishment in spruce-dominated forest typically occurred within 3–7 years of fire. Additional study of the East Fork burn is necessary to determine the proportion of sites that were conifer before the burn and grass after the burn that may subsequently transition to shrub or hardwood establishment from seed by some future date (see Conclusions and Recommendations).

We purchased Quickbird imagery during the first summer Digital Globe acquired scenes in Alaska. The high resolution of Quickbird was useful for geo-referencing raw imagery because rocks and individual trees (live crown or shadow) were visible for ground validation by GPS. The image discerned 3–4 m diameter patches of grass and herbaceous species such as fireweed (*Epilobium angustifolium*). However, one tradeoff of higher resolution imagery is that the off-nadir angle of acquisition detects subtle shadows (particularly on taller objects) that complicate segmentation choices for constructing polygons. Data storage and software capabilities to handle large files are also important considerations as imagery resolution increases. We chose a photography scale of roughly 1:12,000 because it provides high resolution, but it resulted in large file size (900 MB per frame). Similarly, the file size of the September 2002 Quickbird image fused with panchromatic was 7.0 GB at 16 bit or 3.5 MB at 8 bit. To improve processing time, we divided the image (272 km^2) into 9 parts for analysis because Version 3.0 of eCognition was limited to 2 million polygons.

Another factor in our choosing Quickbird imagery was the potential with high resolution to distinguish patches of willow and other woody species preferred as browse by moose from nonpreferred species such as alder. Groesz and Kastdalen (2007) achieved 78% accuracy to a similarly coarse classification as ours (pine, spruce, deciduous, and other groundcover) when Quickbird imagery and eCognition was used to classify shrub and tree polygons in boreal forest of Norway. In that study area, moose browsing had reduced the subcanopy (difficult to detect using overhead sensing), reduced biomass, or affected plant growth form. Availability of type classifications produced from Quickbird and eCognition for forest management purposes near Fairbanks (e.g., hazardous fuels, timber inventory) may allow subsequent testing of accuracy with easier access to specific stands. However, any patterns detected near Fairbanks (ca. 100–500 m elevation) should be validated in the forest types of our study area, which is at a higher elevation that includes treeline and alpine. Groesz and Kastdalen (2007) recommended hyperspectral imagery to improve browse species distinction. The ability to map smaller stands of browse species would increase the required sample size of validation points at specific type locations to determine if the spectral signatures are different enough for consistent typing across potentially high landscape variability.

Satellite imagery and aerial photography each have practical advantages and constraints. Imagery provides seamless digital coverage of large areas in 4 color bands that can be orthorectified with a few GCPs. We had desired acquisition of the imagery during 15 July–15 August to correspond to pre-burn CIR photos for change detection during similar vegetation phenology. However, tolerances for cloud cover (≤20%) and competing orders delayed successful Quickbird acquisition until mid-September, after vegetative senescence and substantial leaf fall. Aerial photography should have a reduced effect from atmospheric contamination by natural and anthropogenic sources because of a shorter path from sensor to target than satellite imaging. Also, local flight crews can readily assemble in response to good conditions at specific locations. Disadvantages to high-resolution photography include only 3 color bands and a smaller image footprint that requires several photos to assemble a mosaic. Base maps are commonly absent in remote regions of Alaska, so each photo requires GCPs. Producing a mosaic from flight lines overlapping in opposite directions can be challenging because of solar angle and shadows.

Our original intent to compare the labor cost of classification between aerial photos and imagery was hindered by the GIS staff turnover at DOF during the course of the project (eCognition software has a steep learning curve). Regarding material costs, the digitized CIR photos cost $\$3.25/km^2$ without overlap or $\$21/km^2$ with 60% endlap and 30% sidelap for stereoscopic viewing, whereas the Quickbird imagery was $\$30/km^2$, including the ephemeris data needed for image processing. The smaller Quickbird scene from July 2002 was $\$24/km^2$ because it was a public archive. Our use of existing equipment (camera and aircraft) to produce film negatives that were digitally scanned likely compromised resolution compared to contracting for digital photography; however, the latter would be a substantially greater expense (estimated at $\$120/km^2$ for 4 m multispectral/0.6 m pan with GCPs). Technology associated with high resolution imagery or digital photography is changing rapidly, so many of our challenges a decade ago with orthorectification and image processing may now be resolved.

Conclusions and Recommendations

The 1998 prescribed burn was a comparatively inexpensive treatment per unit area that increased shrub and deciduous sapling cover from 15% to 53% after 4 growing seasons in the 25 km^2 study area. Our modeling of seral transition from woodland spruce to shrub and deciduous sapling indicated a preference for transitioning to shrub rather than grass at lower elevations and steeper slopes for all but extreme (≤5% of area) elevation and slope combinations found in the study area. We caution readers that experimentally the East Fork burn represented a single treatment under specific fire conditions (fuel composition and moisture, weather during ignition, fire intensity, and fire severity), and we were not able to include fire severity in our analysis of factors associated with seral type conversion.

Our initial post-burn classification for change in shrub prevalence after 4 growing seasons was primarily on sprouts but provides a baseline for future change detections. Future evaluations of this burn should obtain more ground validation to build a more robust error matrix for classification accuracy for the goal of determining what proportion of initial grass regeneration on sites that were conifer before the burn becomes shrub or hardwood following establishment from seed after another 10–20 years. Establishment of willows and hardwoods from seed subsequent to this initial evaluation should be possible to detect in future remote sensing. We expect these species attractive to moose as winter browse to appear mostly in areas classified as grass in the 2002 image, although the chance of establishment declines markedly after the first few years post-fire (Johnstone et al. 2004). Relatively few of the small (≤15 cm dbh) black spruce snags that dominated the study site were wind-thrown by 2005, but over time exposure of mineral soil at root wads of fallen snags may provide germination strata on microsites for willows. A site visit to the study area prior to undertaking a subsequent evaluation is recommended to verify individual shrubs or young trees grown from seeds (as opposed to sprouts) by pulling some sample plants to examine the root structure.

Spatial analysis of seed sources may provide an additional covariate to improve modeling of the potential for conifer to shrub transition over time. Distance of seed source from recently burned sites may influence potential for seral conversion from conifer to shrub or hardwood sapling. Seeds of the willow family (Salicaceae) have cottony tufts and minute seed capsules for buoyancy that aid primary dispersal by wind and secondary dispersal by water. Aspen seeds can be carried for several kilometers by air currents and remain viable to germinate for 2–4 weeks with adequate moisture (Burns and Honkala 1990). Willow seeds may remain viable for days to weeks depending on species (Densmore and Zasada 1983). Most willow seeds in riparian communities tend to be entrapped on substrate within a few hundred meters from the source (Gage and Cooper 2005); successful dispersal distance for willows on upland sites is unknown. A question for assessing temporal value of this burn to moose as winter forage is what proportion of the present "shrub" class is shrub-form willow that may remain largely within reach of moose as winter forage compared with nascent hardwood forest that may grow beyond reach of moose in a few years.

Our fire treatment conditions should be replicated in a similar suite of vegetation types to verify whether a similar response pattern occurs and should include other explanatory variables that influence seral conversion. Further experimentation with prescribed fire in similar vegetation types but under different fire conditions would be beneficial to see if the initial seral type changes we observed on the East Fork burn are consistent in other areas. For future studies we recommend conducting a pre-burn type classification before the burn with remote sensing data of similar type and resolution to data planned for use in the post-burn assessment of burn severity and type classification to reduce complications in analysis. The high resolution of Quickbird imagery has been useful to DOF for classifying timber and fuel types through object-oriented analysis, but the creation of unique polygons between pre- and post-burn classifications required dropping analysis back to the pixel level for pre- and post-burn comparison, thus greatly increasing computational demands and constraining certain analyses. The resolution of terrain metrics was far coarser than the vegetation types, as would likely be a fire severity classification. To facilitate spatial analyses (such as distance to nearest seed source as a factor in seral type conversion), future studies might use imagery of lower resolution (e.g., 5 m) for a reduced size of the digital data set, particularly if a larger study area is desired.

Study sites closer to settlements would be easier access for validating classifications of burn severity (ideally within 1 year of the burn) and vegetation type, and the existence of a orthorectified base map would further reduce spatial validation costs. As of April 2015 the federal effort on monitoring trends in burn severity (www.mtbs.gov) has provided dNBR and related parameters for 893 wildland and prescribed fires in Alaska, although the East Fork prescribed burn was not among those evaluated. Field validation of estimated severity parameters using the composite burn index (Keys and Benson 2006) within 1–2 growing seasons after the burn is advisable. The standard composite burn index did not perform well for Alaska conditions (Kasischke et al. 2008), so we recommend a modied methodology developed for Alaska (Alaska Interagency Fire Effects Task Group 2007) and later modified by Barrett et al. (2010). Study sites closer to settlements also might have spatial data on soil moisture, which influence fire severity (Dyrness and Norum 1983). From a management perspective, burn sites closer to settlements may also have better access by hunters to utilize any gains in moose abundance caused by improved habitat. However, conducting a prescribed burn closer to settlements is more challenging with respect to social concerns over smoke and property (Paragi et al. 2009).

We studied vegetation change at the landscape scale to understand the potential for improving winter range for moose and to understand selected factors of the environment (pre-burn vegetation and terrain metrics) that may be useful in burn prescriptions to enhance shrub and hardwood regeneration in spruce-dominated boreal forest. Johnstone et al. (2008) provided a key for predicting post-fire succession in black spruce communities that may be helpful in writing a burn prescription. An alternative technique for directly assessing improvements in moose winter range on a prescribed burn is to obtain pre-treatment estimates of browse production and utilization (Seaton 2002, Paragi et al. 2015) and evaluate subsequent changes on treatment and control sites. Seaton's technique based on forage biomass was being developed when this project began. The ideal study design to infer fire effects would include browse production and removal before the fire, both within the burn perimeter and in adjacent control areas, for comparison to post-burn data. This type of design is known as before-after-control-impact (Green 1979). An advantage of the browse evaluation technique over vegetation type change with remote sensing data is that browse surveys do not require specialized data processing equipment or analytical expertise outside our agency. Based on our experience with this project, we attempted to use a before-after-control-impact design to evaluate fire effects on moose browse production and removal in subalpine habitat of Game Management Unit 20A, but the burn never occurred because of logistic constraints (Paragi and Kellie 2011:17).

Monitoring response to a burn by selected wildlife species is the next step to evaluation of prescribed fire efficacy. Estimating the abundance, distribution, and productivity of wildlife provides metrics for progress toward or achievement of population and harvest objectives and evaluating nutritional condition, ideally with a before-after-control-impact design in burned and unburned areas. Although moose are generally not known to shift their distribution to previously unused areas as a result of fire, they do spend more time on fire-improved range during seasonal movements (Gasaway and DuBois 1985). We expect that the initial increase in shrub and sapling cover and browse in the study area could support more moose than before the burn if nutritional condition improves for resident moose, although predation could hinder or prevent a numeric response (e.g., Gasaway et al. 1992). Estimates of moose abundance and age-sex composition in Game Management Unit 20E are done annually from fixed-wing aircraft in early winter. Surveys of randomly sampled polygons based on topography (Gasaway et al. 1986) occurred within the East Fork study area before and after the burn. Subsequently a spatial sampling technique based on rectangular sample units (Kellie and DeLong 2006) has included the East Fork study area post-fire. Although the number and location of random sample units within the burn perimeter has not been consistent among years or surveys and location of individual moose was not recorded (C. Gardner, ADF&G, personal communication, 2006), these survey data may provide a means to grossly compare distribution and relative abundance of moose in sampling cells within the study area before and after the burn, ideally compared with a nearby area not recently burned. Patchiness of burned area within cells could complicate this type of analysis if few cells were entirely burned or unburned. If data on twinning rate soon after parturition existed in a similar design context, moose nutritional response could be used to infer fire effects without the confounding effect of predation (Boertje et al. 2007). However, it would be difficult without a sample of radio-marked adult cows to achieve an adequate sample size inside and outside the burn given vegetative cover and relatively low density moose in this area (Gardner 1998, Gross 2006). Known distribution of radio-marked cows could also permit weights of 10-month-old calves to infer nutritional condition (Boertje et al. 2007) if there were adequate samples inside and outside the burn, before and after the burn.

Where pre-treatment data are lacking, comparison between burned and adjacent unburned areas is often the default option to assess fire effects on wildlife abundance and distribution. In the post hoc design, it would have to be assumed the adjacent forest is of similar composition to the pre-burn condition, recognizing that a fire perimeter and proportion of area that actually burns within the perimeter is determined in part by differences in soil moisture and vegetation type (thus potentially different community from the burned area) and in part by fire weather conditions that influenced spread of a fire (e.g., Eberhart and Woodard 1987). Use of aircraft to conduct track counts of forest carnivores such as martens (*Martes americana*) or lynx (*Lynx canadensis*) or major prey such as snowshoe hares (*Lepus americanus*) may be feasible for comparing habitat selection between the burn and adjacent older forest (Paragi et al. 1996, 1997), with the limitations of a post hoc study design. During field visits in the East Fork burn we noted the diggings of yellow-cheeked voles (*Microtus xanthagnathus*) in burned areas. These voles are a preferred prey of martens in burns, so we expect marten abundance may have increased in the first decade after the burn because of fire effects on prey and habitat features such as coarse woody debris (Paragi et al. 1996).

Demonstrating an increase in moose harvest to a prescribed fire is the ultimate policy evaluation for intensive management because any functional or numeric response of moose to burns must then be accessible to hunters (Brinkman et al. 2013). Moose harvest is reported by large drainage scale in this part of Alaska (4,032 km^2 for the prescribed fire area, 275–2,953 km^2 for adjacent reporting areas), so relating harvest to this specific burn that lacks trail or motorboat access may be difficult. Patterns of moose movements and the effect of large adjacent wildland fires would also complicate inference on harvest from this prescribed burn. Future habitat enhancement projects should incorporate spatial and temporal facets of wildlife and harvest monitoring into design scoping.

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APPENDIX. Poster presented at International Boreal Forest Research Association 12th Annual Scientific Conference, 3–6 May 2004, Fairbanks, Alaska.