ANALYSIS PAPER

Change in Vegetation Productivity for Three National Forests in Utah, 1986-2011: Dixie, Fishlake, and Manti-La Sal National Forests

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Background

Ongoing climate change will substantially alter the ecological communities of the Colorado Plateau with warming temperatures, shifting precipitation patterns, and increasing drought severity projected to continue for the Southwest (Garfin et al. 2013). Projected climate changes will interact with existing disturbances on the landscape, and species and ecosystems will be affected in diverse ways. As these climatic shifts increase plant stress and mortality, vegetation distributions will change considerably through the century (Rehfeldt et al. 2006, Bureau of Land Management 2012).

Vegetation productivity (i.e., primary productivity) is influenced not only by soil characteristics, ecological community composition, herbivory, and disturbance (Milton et al. 1994, Field et al. 1995), but also by climatic shifts (Breshears et al. 2005, Schwinning et al. 2008, Reeves et al. 2014). Anthropogenic land use can compound the effects on vegetation

of a changing climate (DeFries et al. 1999). This presents a formidable challenge to public land managers who strive to effectively balance multiple land uses with ecosystem health to support the long-term sustainability of both. Monitoring indicators of ecosystem health, such as primary productivity, can aid in identifying areas sensitive to disturbance and can aid land management strategies that mitigate further environmental degradation.

Primary productivity of terrestrial vegetation is an important indicator of ecosystem health (McArthur et al. 2000, Hunt et al. 2003). Areas with vegetation that sustains high productivity during low productivity (i.e., drought) periods may also provide climate refugia for species (Klein et al. 2009) and contribute to the adaptive capacity of an ecosystem impacted by climate change (Mackey et al. 2012, Gould et al. 2015).

Although vegetation productivity is directly related to photosynthesis and is an important metric of ecosystem dynamics, direct estimates of vegetation productivity are laborious to measure on the ground (Gower et al. 1999) and are subject to a high degree of variability dependent upon location, frequency, and assessment methodology (Lauenroth et al. 2006). The increasing availability of remotely-sensed data and derived vegetation indices (e.g., Normalized Difference Vegetation Index [NDVI], Soil Adjusted Vegetation Index [SAVI], and the tasseled cap indices) have made landscape-level estimates of surrogates for vegetation productivity possible, enabling ecologists and land managers to assess trends in ecosystem condition and response to environmental changes (Pettorelli et al. 2005). Specifically, variation in NDVI has been shown to be a function of complex relationships among climatic, topographic, and latitudinal zonation factors that impact vegetation growth (Ramsey et al. 1995). Metrics such as the interannual change in NDVI can reflect variation in climate in areas without disturbance (Hargrove et al. 2009, Norman and Hargrove 2011) and can help land managers highlight landscape-scale climate sensitivities.

The National Forests in Utah, including the Dixie, Fishlake, and Manti-La Sal, will not be exempt from the effects of climate change; thus, their land managers are faced with difficult but important decisions on how best to maintain the landscape subject to multiple uses. We present a 26-year (1986-2011) analysis of productivity across the three National Forests to identify those areas where declines in productivity have occurred as one contribution to

understanding climate vulnerability in these important landscapes. We specifically looked at how this change in vegetation productivity varied by vegetation type to identify vegetation types in which primary production declines may have implications for land management.

Methods

Vegetation Productivity

To evaluate the change in vegetation productivity over the study period in the three National Forests, we analyzed publicly available, vegetation-specific satellite imagery in a Geographic Information System (GIS) and in the R programming environment (R Core Team 2014). Specifically, we used LANDSAT 5 Thematic Mapper (TM) data available for the years 1986-2011 (obtained from the U.S. Geological Survey's GloVis data distribution site: glovis.usgs.gov) for the full extent of the three National Forests (total study area: 20,809 km^2). This type of satellite data can be used to represent net primary productivity of vegetation through the metric of Normalized Difference Vegetation Index (NDVI), a robust tool for assessing ecological responses to environmental change (Pettorelli et al. 2005).

For the 1986-2011 time period (which encompassed all possible years of available data for LANDSAT 5 TM), one set of images was obtained each year during the month of June, a month reflecting summer vegetation productivity prior to later-summer, monsoon rains. Due to a limited availability of quality images each month and the need for eight images in each set (i.e., a set consists of eight images each year to cover the extent of the study area), some years included the next best image which was selected between April and July. Images were pre-processed for analysis (i.e., orthorectified, clouds removed, and converted to at-sensor radiance) using the *landsat* package in R. Because changes in atmospheric conditions (e.g., increased humidity, smoke, regional haze) can affect estimates of NDVI, we calculated atmospherically-corrected surface reflectance values using the dark-object subtraction method (Chavez 1989) also implemented within the *landsat* package. NDVI values were then calculated according to this standard formula:

$$NDVI = \frac{(Band\ 4 - Band\ 3)}{(Band\ 4 + Band\ 3)}$$

Results were determined using the *raster* package in R. To calculate the change in vegetation productivity during the study period, we used the difference between a 10-year average of NDVI for the 1986-1995 time period (historical vegetation productivity) and the 2002-2011 time period (recent vegetation productivity). These two 10-year averages were used to help reduce the effect of year-to-year variation and obtain a more robust historical trend.

Vegetation Type

Because changes in vegetation productivity were not homogeneous across the three National Forests, we stratified our results by vegetation type to identify vegetation types in which productivity may have changed more than others. We used publicly available satellite data of the most recent existing vegetation type from LANDFIRE (2010; www.landfire.gov), a 30-m pixel resolution, and a simplified version of the Society of American Foresters and Society for Range Management (SAF_SRM) vegetation classification[2] as described by Shiflet (1994). For each National Forest, we mapped the change in vegetation productivity within the GIS. We then calculated and graphed the average change in vegetation productivity by vegetation type for each National Forest.

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[2] The following descriptions indicate how the original SAF_SRM classification was consolidated: Aspen (SAF 217: Aspen); Deciduous Shrubland or Chaparral (SRM 413: Gambel oak, SRM 415: Curlleaf Mountain-Mahogany, SRM 421: Chokecherry-Serviceberry-Rose, SRM 503: Arizona Chaparral); Desert Scrub or Shrubland (SRM 212: Blackbush, SRM 414: Salt Desert Shrub, SRM 501: Saltbush-Greasewood, SRM 506: Creosotebush-Bursage, SRM 605: Sandsage Prairie); Developed (LF 20: Developed, LF 80: Agriculture); Forb or Grassland (SRM 106: Bluegrass Scabland, SRM 409: Tall Forb, SRM 410: Alpine Rangeland, SRM 502: Grama-Galetta); Introduced Riparian Vegetation (LF 58: Introduced Woody Wetlands and Riparian Vegetation); Introduced Upland Vegetation (LF 54: Introduced Upland Vegetation – Herbaceous); Mixed Conifer (SAF 206: Engelmann Spruce-Subalpine Fir, SAF 209: Bristlecone Pine, SAF 210: Interior Douglas-Fir, SAF 211: White Fir, SAF 218: Lodgepole Pine, SAF 219: Limber Pine, SAF 237: Interior Ponderosa Pine); Pinyon-Juniper Woodland (SRM 412: Juniper-Pinyon Woodland, SRM 504: Juniper-Pinyon Pine Woodland); Riparian (SAF 235: Cottonwood-Willow, SRM 203: Riparian Woodland, SRM 418: Bigtooth Maple, SRM 422: Riparian); Sagebrush (SRM 314: Big Sagebrush-Bluebunch Wheatgrass, SRM 402: Mountain Big Sagebrush, SRM 403: Wyoming Big Sagebrush, SRM 405: Black Sagebrush, SRM 406: Low Sagebrush); Sparse (LF 33: Sparsely Vegetated, No Dominant Lifeform, Non-vegetated).

Results

Vegetation Productivity across the National Forests

Across the three National Forests, we detected both negative NDVI values (declines) and positive NDVI values (increases) in change in vegetation productivity in particular areas in the Dixie (Figure 1), Fishlake (Figure 2), and Manti-La Sal (Figure 3) for the 1986-2011 study period. However, the average vegetation productivity was a negative value for each forest (decline; Figure 4).

Vegetation Productivity across Vegetation Types

The dominant vegetation types (in terms of area) differed across each of the three National Forests (Figure 5). For the Dixie, pinyon-juniper woodland (33% of area), mixed conifer (21% of area), and aspen (16% of area) were the most prevalent. For the Fishlake, pinyon-juniper woodland (30% of area), aspen (19% of area), and sagebrush (14% of area) were the most prevalent. For the Manti-La Sal, aspen (26% of area), pinyon-juniper woodland (25% of area), and mixed conifer (17% of area) were most prevalent.

All vegetation types in the three National Forests showed an average negative value for our change in NDVI calculation, indicating a decline in vegetation productivity (Figure 6). In the Dixie National Forest, the forb or grassland, mixed conifer, and aspen vegetation types had the largest declines, respectively. In the Fishlake National Forest, the mixed conifer, introduced riparian vegetation, and riparian vegetation types had the largest declines, respectively. For the Manti-La Sal National Forest, the mixed conifer, pinyon-juniper woodland, and introduced riparian vegetation types had the largest declines, respectively.

Discussion

Vegetation Productivity across the National Forests

We detected an average decline in vegetation productivity over a recent 26-year period across three National Forests in Utah. This average decline in vegetation productivity across vegetation types within the three National Forests may foreshadow the continued decline in

vegetation productivity projected for much of the Southwest's grazed lands over the rest of the century (Reeves et al. 2014).

We did not explicitly model relationships between vegetation productivity and livestock grazing or climate variables (e.g., historic precipitation or temperature records) and this analysis cannot attribute the average decline in vegetation productivity to one specific cause. However, additional declines in vegetation productivity can be expected with ongoing climate change, rendering ecosystems increasingly vulnerable to other disturbances (Allen et al. 2010, Garfin et al. 2013, Seager et al. 2013, Gaitán et al. 2014). This will likely necessitate the re-evaluation of land uses (e.g., livestock grazing, logging, recreation) that can compound impacts from other disturbances such as wildfire, non-native species invasions, and forest pest outbreaks.

Livestock grazing is one such land use and occurs across nearly all the Dixie, Fishlake, and Manti-La Sal National Forests. Reduced average vegetation productivity is a red flag for ecosystem health and suggests that the landscape may not be able to sustain livestock grazing at the current number or intensities. Livestock grazing has a widespread influence on native ecosystems in the region and has the potential to exacerbate climate change impacts on the landscape (Beschta et al. 2013, 2014, Svejcar et al. 2014). While this influence varies by livestock management practice and ecosystem type, numerous studies have identified ecological impacts such as altered vegetation diversity (as summarized by Fleischner 1994) and disrupted biological soil crusts (Belnap et al. 2006, Schwinning et al. 2008), which has been linked to nonnative species spread (Guenther et al. 2004, Reisner et al. 2013). We suggest that land managers take steps to reduce compounding anthropogenic impacts such as intense livestock grazing to improve the ability of the landscape to resist or adapt to the pressures of climate change.

Vegetation Productivity across Vegetation Types

Our average change in vegetation productivity calculations detected declines in vegetation productivity in all vegetation types, and the largest declines were detected in aspen, forb or grassland, pinyon-juniper woodland, mixed conifer, riparian, and introduced riparian vegetation types.

Importantly, our analysis was not designed to account for or to determine changes in vegetation type over the study period. Vegetation types within the three National Forests may have changed since 1986, shifting in extent, seral stage, and/or dominant vegetation cover, but the frequency of available LANDFIRE existing vegetation type data did not allow for such changes to be tracked during the study period. For example, areas classified as introduced riparian or introduced upland vegetation may have represented native riparian or native upland vegetation prior to 2010 – an important consideration for land managers. While these shifts do not directly impact our results, understanding shifts in vegetation type over time aids in further characterizing the management implications associated with increases or declines in vegetation productivity. As vegetation type is another important indicator of health of grazed lands (Hunt et al. 2003), exploration of these shifts during recent decades could be a useful expansion of this assessment.

Aspen Vegetation Type

Our results for a decline in aspen vegetation productivity coincide with the widely-documented climate-driven decline in aspen (*Populus tremuloides*) forests across much of the American West (Worrall et al. 2013). In Utah specifically, documented declines of aspen from 2000-2004 and in 2007 as determined by NDVI have been related to extreme drought conditions compounded by shifts in seasonal frost phenology (Currit and St Clair 2010). Additional pressure from chronic ungulate browsing on aspen ramets (Seager et al. 2013) has been a compounding effect on long-term aspen decline (Worrall et al. 2013). Wild and domestic ungulate browsing have limited aspen recruitment in south-central Utah, including in the Dixie and Fishlake National Forests (Kay and Bartos 2000). The impacts of wild and domestic ungulate browsing on aspen recruitment continue to be a concern for aspen in the western United States (Rogers and Mittanck 2014), particularly as they reduce aspen resilience to current and future climate change (Seager et al. 2013). The distribution of aspen is anticipated to decline greatly in the western United States by the end of the century (Rehfeldt et al. 2009).

Forb or Grassland Vegetation Type

Our analysis also detected declines in the forb and grassland vegetation type, although spatial variation was high, particularly for the Dixie National Forest (Figure 6). These results differ from an NDVI vegetation productivity assessment of the Western United States for a similar time period, 1987-2007, which detected an overall increase in grassland vegetation productivity despite later declines (Zhang et al. 2010). The declines we detected in the forb or grassland vegetation type may coincide with the loss of native perennial grasslands to shrub encroachment and non-native grasses in the Southwest over the last century (Gori and Enquist 2003, Briggs et al. 2005, USFS 2014*a*). A repeat photography study in the Fishlake National Forest identified that, of 321 photosets within grassland vegetation types, 39% depicted declines in grassland cover mostly attributable to sagebrush, pinyon-juniper, and other conifer encroachment while in 4% of the photosets grassland cover increased due to shrubland treatment and management (Kay 2003). A similar study in Utah's Henry Mountains identified that, of 152 photosets, 42% depicted declines in grassland cover while in 40% grasslands increased where treatments had removed woody species (Kay 2015)

Our 2010 existing vegetation type data does not account for vegetation type change since 1986, so transitions from grassland to shrubland during this period are unaccountable. Despite the drivers of this vegetation type's decline in productivity, reductions in primary productivity in grassland areas have implications for forage availability for domestic and wild herbivores on the National Forests. Furthermore, herbivory of vegetation during periods of limited productivity can impact the replenishment of ecological reserves (e.g., soil organic matter) which enable vegetation response to rainfall (Noy-Meir 1973, Schwinning et al. 2008). During drought periods, loss of ecological reserves can experience negative feedback, subsequently impacting post-drought vegetation productivity and resilience to disturbance (Schwinning et al. 2008). Specific to domestic herbivory, livestock grazing in low productivity, drought-stricken areas can lead to shifts in vegetation community composition (Milchunas and Lauenroth 1993, Harris and Asner 2003) or invasive species spread (Loeser et al. 2007). For example, a decline in vegetation quality has been predicted in other semi-arid ecosystems due to both increases in aridity and grazing that selects for annual species (DeBello et al. 2005, Hulme 2005). Projected intensifying drought suggests future

reductions in grassland productivity as grass cover is positively related to average annual precipitation (Gremer et al. 2015), particularly in arid regions such as the Southwest (Sohoulande Djebou et al. 2015).

Mixed Conifer Vegetation Type

Our findings of declines in vegetation productivity across the mixed conifer vegetation type reflect other studies of conifer mortality in the last two decades. A study of change in NDVI for the western United States for 2000-2006 found an overall die-off for evergreen species (Yuhas and Scuderi 2009). Our findings may also reflect declines from the heavy impact of bark beetle outbreak on mixed conifer forests across the western United States. Between 1997 and 2009 almost two million acres of Utah's mixed conifer forest experienced mortality due to bark beetles (USFS 2011). This outbreak is expected to continue through the end of the decade (USFS 2011) and can intensify with a warming climate (Logan et al. 2003) and drought conditions (Desprez-Loustau et al. 2006) which can also increase mixed conifer mortality (Guarín and Taylor 2005). Declines in some other conifer species are also projected with climate change, including ponderosa pine (*Pinus ponderosa*) and Douglas fir (*Pseudotsuga menziesii*; Rehfeldt et al. 2006, Bureau of Land Management 2012). It is expected that the most dramatic changes in forested ecosystems amid climate change will materialize through changing disturbance regimes, namely wildfire (Dale et al. 2001). The compounding impacts of wildfire with other disturbances such as ungulate grazing are complex, although post-fire grazing with little rest period (e.g., two years) may reduce overall native grass cover and increase non-native species occurrence (Mork 2010). In forest understories, grazing can increase the susceptibility of grasses to drought (Kolb 2000).

Pinyon-Juniper Vegetation Type

Pinyon-juniper woodlands are considered to be expanding across much of the western United States, particularly into shrub-steppe or sagebrush-steppe vegetation types (Miller et al. 2008), making the decline in vegetation productivity in this vegetation type a potentially counterintuitive result. Our 2010 existing vegetation type data did not account for vegetation type change since 1986, so areas that may have transitioned from grassland vegetation to pinyon-juniper vegetation over the study period (i.e., classified as pinyon-juniper vegetation

in 2010) may be included. While this classification may have complicated the extent to which the productivity of grassland and pinyon-juniper vegetation types have declined, it would not have impacted the overall average decline in vegetation productivity across the three National Forests.

Moreover, pinyon pine (*Pinus edulis*) and juniper (*Juniperus* spp.) can experience different drought-related mortality rates due to their differing water regulation mechanisms (McDowell et al. 2008). Distributions for each of these two species are projected to gradually diverge with climate change (Rehfeldt et al. 2006). As pure stands of pinyon pine are not typically considered a separate vegetation type (Shaw et al. 2005), the extensive mortality in pinyon pine due to drought and interacting pest and disease outbreaks between 2000-2004 (Shaw et al. 2005) give more on-the-ground relevance to our results.

Riparian and Introduced Riparian Vegetation Types

Our calculations detected declines in both riparian and introduced riparian vegetation types. Studies have described the decline of native riparian plants in the Southwest due to changing hydrological regimes that hinder native plant recruitment and amplify invasive species spread (Rood and Mahoney 1990). Native riparian trees, especially cottonwood (*Populus fremontii*) have been considered to be on the decline (Rood and Mahoney 1990, Busch and Smith 1995, Obedzinski and Shaw 2001). Conversely, the now-dominant riparian invasive species of tamarisk (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) have continued to expand across the Southwest over the last several decades (Nagler et al. 2011), adding to other riparian habitat stresses such as drought periods and manipulated flow regimes (Glenn and Nagler 2005).

Long-term riparian vegetation monitoring across the Fishlake and Manti-La Sal National Forests is sparse (USFS 2014*b*). On the Dixie National Forest riparian condition is evaluated based on metrics of herbivory (e.g., stubble height) but excludes measures of streambank stability and stream channelization, limiting the assessment of this vegetation type (USFS 2014*b*). As our existing vegetation type data from 2010 did not capture vegetation changes that may have taken place from 1986-2011, invasive riparian species may have become

more prevalent, suggesting an underestimation of the decline in productivity for native riparian areas.

Natural waters are considered to be hotspots of biological diversity, particularly in arid landscapes. Threats to riparian areas are numerous (Poff et al. 2011). Increased aridity may lead to a reduction in water availability in part due to a grazing-induced increase in soil moisture loss (Landsberg et al. 2002, Hulme 2005). Across the three National Forests, adverse impacts to riparian areas from dams and water diversions, wildfire, roads, and livestock grazing have been documented (USFS 2014*b*). As degraded habitats are typically more susceptible to disturbance, efforts to reduce the impact of non-climatic disturbances such as livestock grazing in riparian vegetation types can reduce climate change vulnerability. Riparian vegetation is rare in the three National Forests (Figure 5) and should be a primary focus for maintaining ecosystem health.

Conclusion

Across the vegetation types of the Dixie, Fishlake, and Manti-La Sal National Forests, vegetation productivity has declined on average over a recent 26-year period. This should be a cause for concern among land managers. This decline in productivity may be attributable to any of several interacting anthropogenic and natural disturbances. Our results, however, suggest that these ecosystems are already stressed and may be increasingly vulnerable to future disturbances. Ongoing and future climate change, disease and pest outbreaks, and shifting wildfire regimes present a challenge for land managers on the three National Forests. To reduce the consequences of further declines in vegetation productivity, managers can reduce the impacts of compounding disturbances through careful land use decisions that prioritize ecosystem health. A loss of vegetation productivity can represent an increased vulnerability to natural disturbances including climate change. Interactions between climate change impacts and multiple-use management present significant challenges to land managers, but they also present an opportunity to balance sustainable land uses with ecological integrity. The decreasing trends we detected in this study encourage future strategic decisions about livestock management on these National Forests

that factor in the impacts of climate change on productivity, particularly in vegetation types that appear to be especially vulnerable to natural disturbances.

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Figures
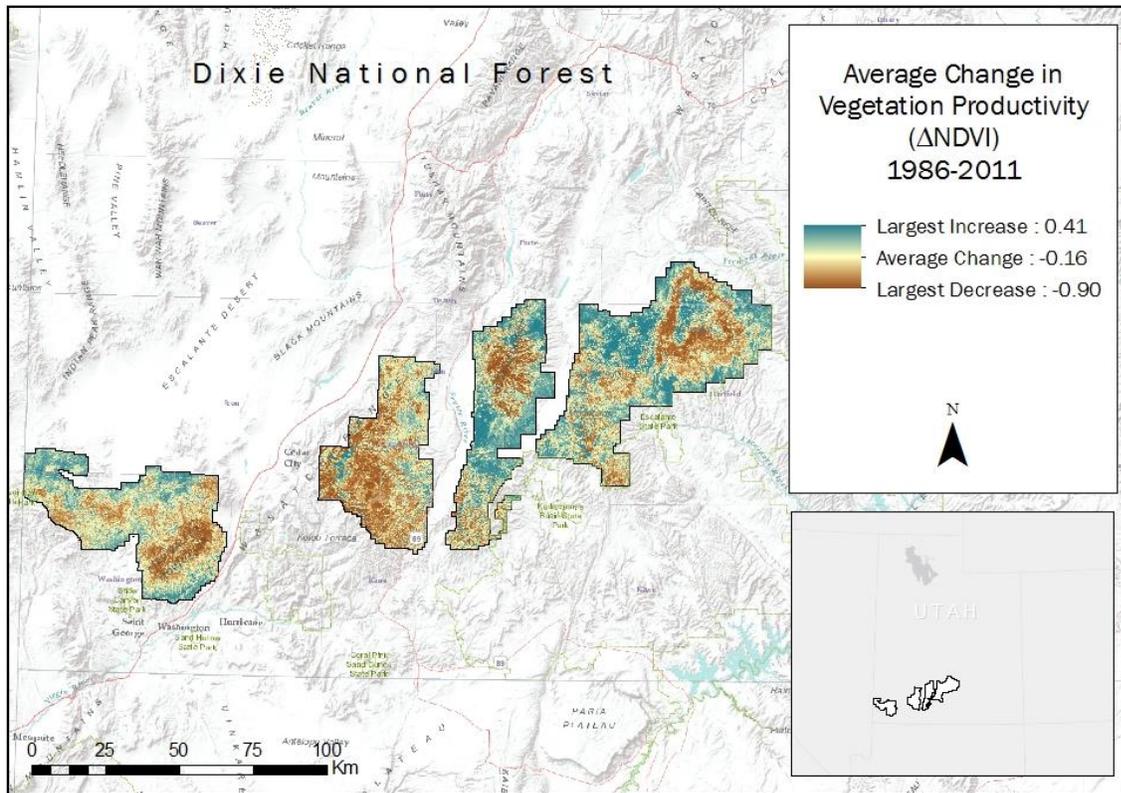
Figure 1 - Average change in vegetation productivity (ΔNDVI) for the Dixie National Forest, 1986-2011. NDVI is a vegetation greenness index and is therefore unitless. It can range from -1 to 1.
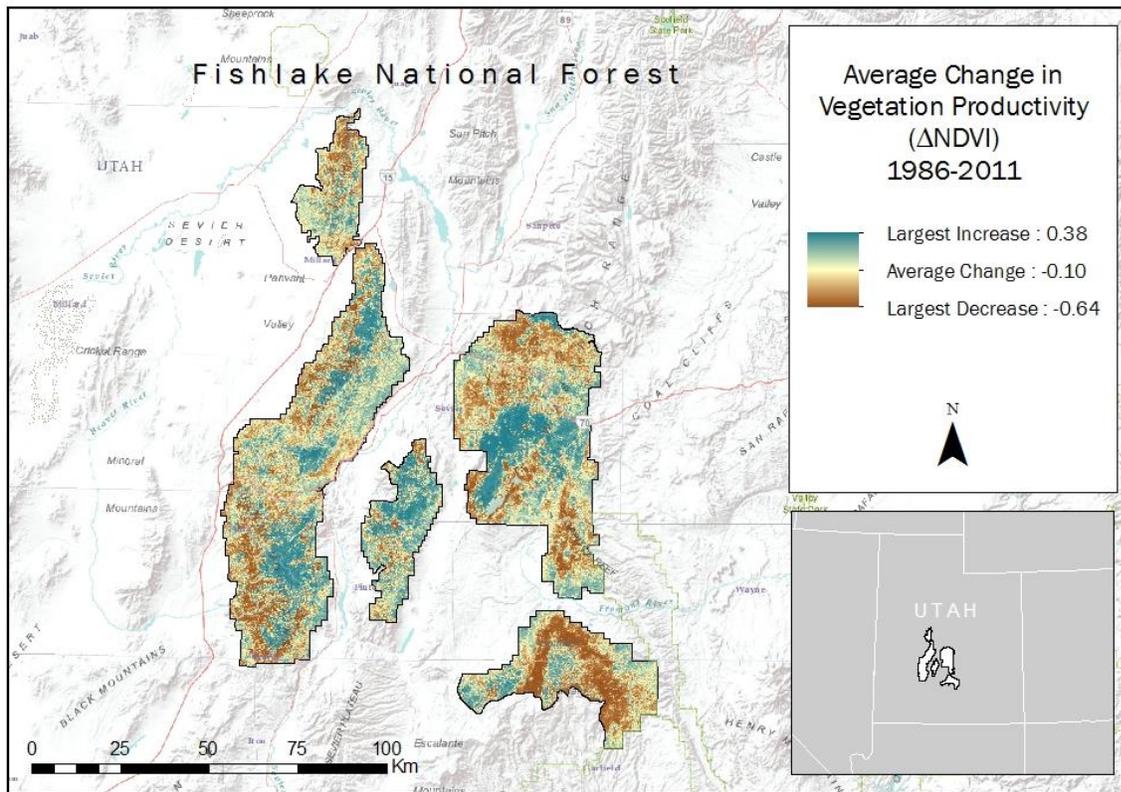
Figure 2 - Average change in vegetation productivity (ΔNDVI) for the Fishlake National Forest, 1986-2011. NDVI is a vegetation greenness index and is therefore unitless. It can range from -1 to 1.
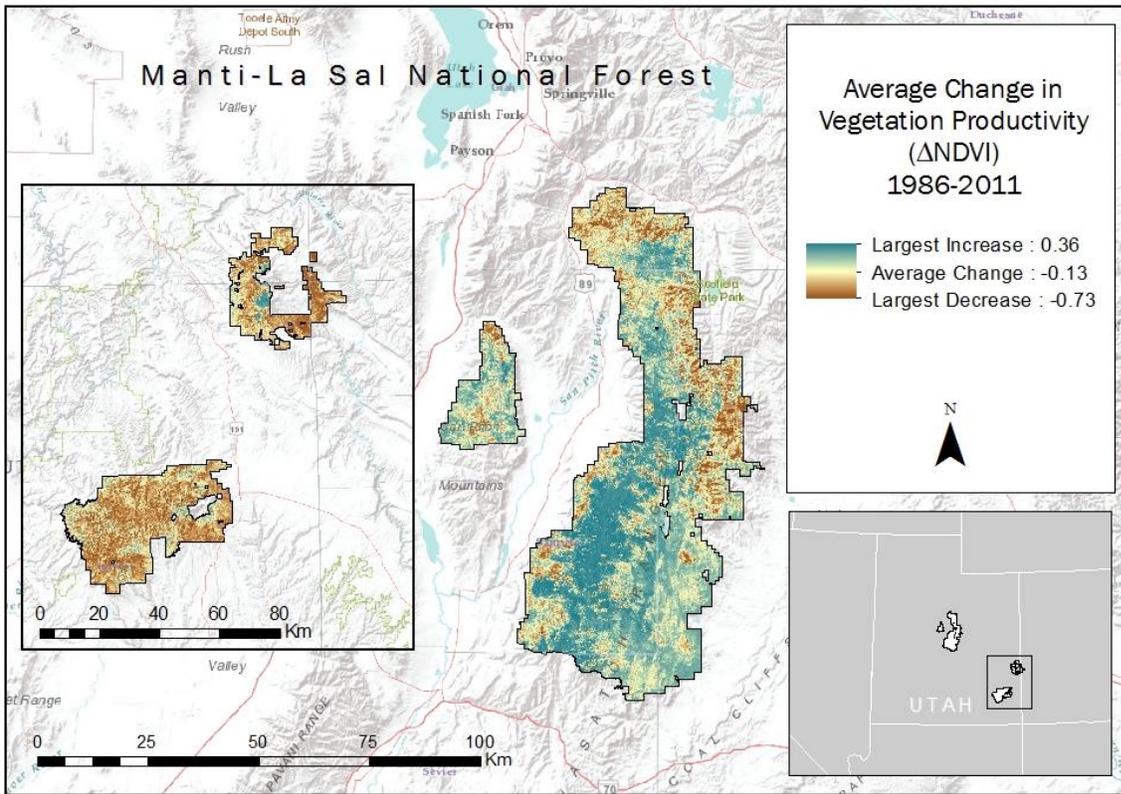
Figure 3 - Average change in vegetation productivity (ΔNDVI) for the Manti-La Sal National Forest, 1986-2011. NDVI is a vegetation greenness index and is therefore unitless. It can range from -1 to 1.
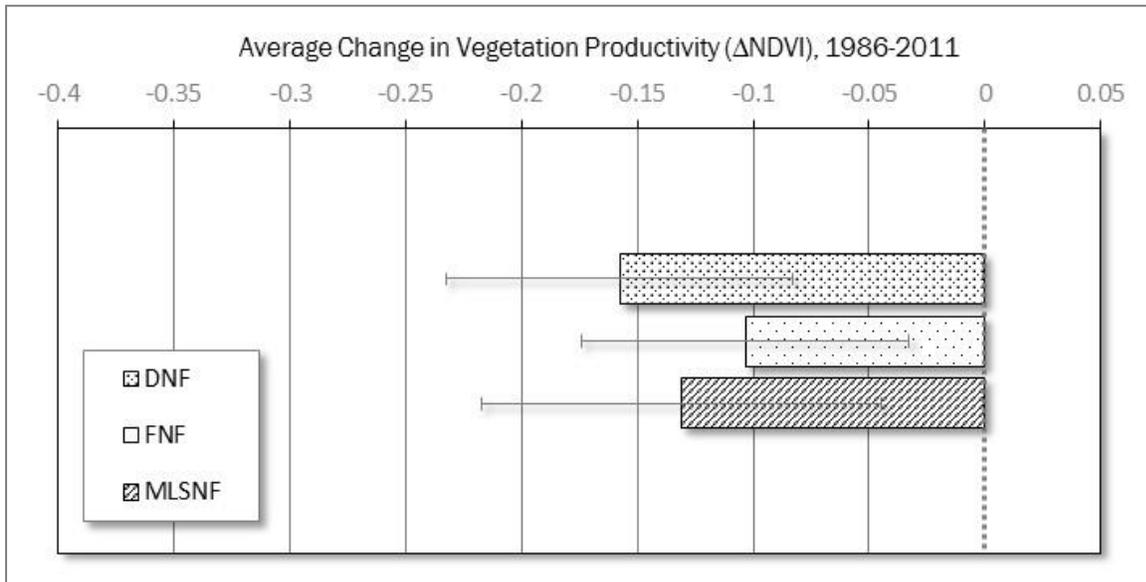
Figure 4 - Average change in vegetation productivity (ΔNDVI) for the Dixie (DNF), Fishlake (FNF), and Manti-La Sal (MLSNF) National Forests for 1986-2011. Although vegetation productivity values ranged from negative (decline) to positive (increase), there was an average decline in vegetation productivity for each of the forests. NDVI is a vegetation greenness index and is therefore unitless. It can range from -1 to 1.
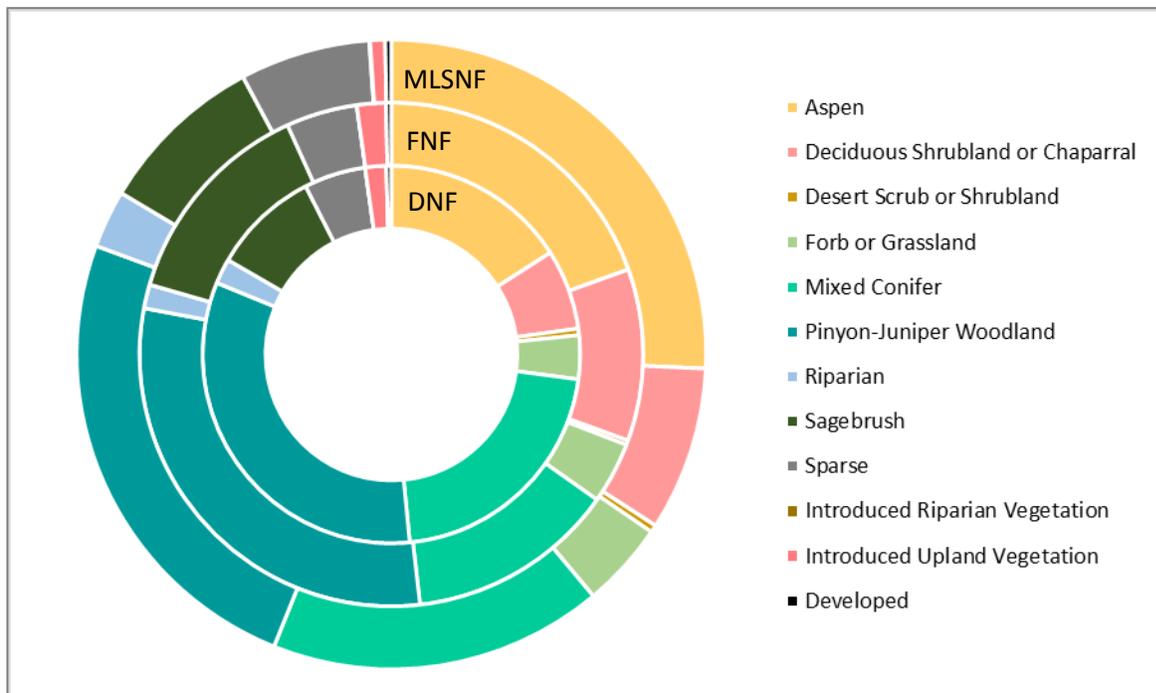
Figure 5 - Portion of National Forest area covered by each vegetation type for the Dixie (DNF), Fishlake (FNF), and Manti-La Sal (MLSNF) National Forests.
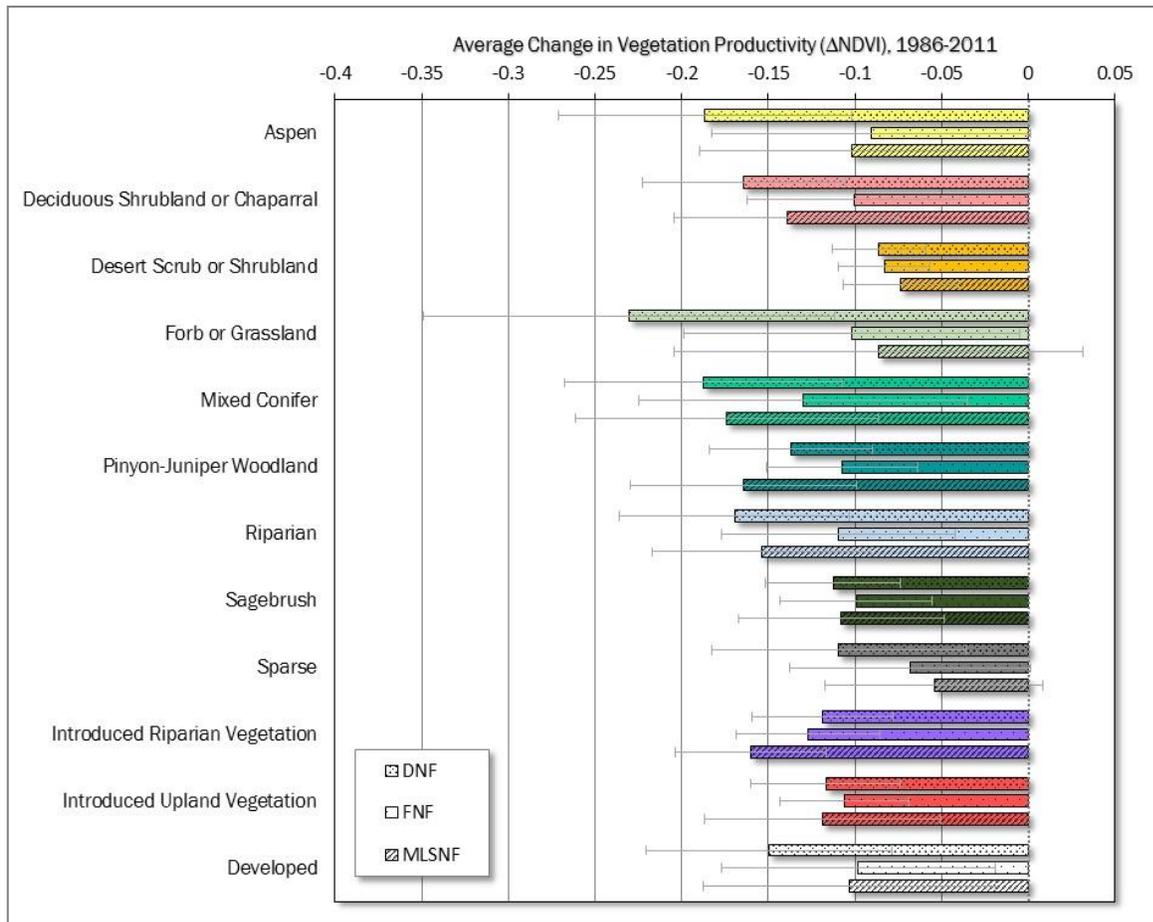
Figure 6 - Average change in vegetation productivity (ΔNDVI), 1986-2011, for the Dixie (DNF), Fishlake (FNF), and Manti-La Sal (MLSNF) National Forests by vegetation type. Although each vegetation type had vegetation productivity values ranging from negative (decline) to positive (increase), the average values for each depicted a decline in productivity. NDVI is a vegetation greenness index and is therefore unitless. It can range from -1 to 1.