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Remote sensing of land-cover change and landscape context of the National Parks: A case study of the Northeast Temperate Network

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ABSTRACT

National park units and protected areas face critical management challenges because of changing land-cover types and variability of landscape contexts within and adjacent the park boundaries. In this study we developed and implemented a multi-scale protocol for detecting and monitoring land-cover change in and adjacent to National Parks and ten segments of the Appalachian National Scenic Trail (AT) in the northeastern United States. We used Landsat imagery from 1970 to 2002 and recent ground-based photography to evaluate changes within park boundaries and within 0.5, 1, and 5 km buffers. The study concluded that all of the studied park units, except one segment of AT in Maine, experienced increases of urban land and declines of forest cover in the immediately adjacent areas and extended buffer zones. Over 30 years and across all parks and trail segments, urban land increased 172% and 181% within 0.5 and 1 km, respectively, of the park boundary or trail centerline. Over the same time period, forested area decreased by 5% and 6% within 0.5 and 1 km, respectively, of the park boundary or trail centerline, with more loss of forest near the parks (18%) than the trail segments (2%). This study provided baseline data demonstrating land-cover alteration over the past three decades and a foundation for a land-cover change and landscape context protocol suitable for monitoring future changes of National Parks and protected areas.

1. Introduction

Suburban sprawl, timber harvests and increasingly fragmented natural habitats are just a few of the factors that impact the ecosystems within and around National Parks and natural reserves (Hansen & Rotella, 2002). Those types of human-induced land-cover change transform natural habitats and pose the single most important threat to biodiversity (Wessels et al., 2004; Sala et al., 2000; Soule, 1991). National park managers across the country are confronted with increasingly complex and challenging issues that require a broad-based understanding of the status and trends of each park's natural resources as a basis for making decisions, working with other agencies, and communicating with the public to protect park natural systems and native species (Fancy et al., 2009; Gross et al., 2006). Land-use legacies can persist for a long time, influencing plant species composition, nutrient cycling, water flows, and climate. Understanding how land use and land cover have affected regional landscape configuration and composition can provide a historical framework for measuring associated changes in ecosystem function and can be used to guide restoration where desirable and feasible (Wilkinson et al., 2008). Knowledge of historical trends of land-cover change, not only *how much* has changed but also *where* and *when* changes have occurred, can help land managers identify key resource and ecosystem stressors, as well as prioritize management efforts (Shriver et al., 2005).

The National Park Service (NPS) Vital Signs Monitoring Program, which is primarily implemented by the NPS Inventory & Monitoring (I&M) networks, aims to monitor a subset of physical, chemical, and biological elements and processes of park ecosystems that are selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values (Fancy et al., 2009). The Northeast Temperate Network (NETN), together with most other NPS I&M networks, has identified "landscape dynamics" as a high-priority vital sign (Mitchell et al., 2006). This vital sign includes the change in area and distribution of ecological systems within and adjacent to parks, extent of major disturbances, and integrity of the ecological systems. Urban development, for example, is one of the most important stressors that the parks and other protected areas are facing. The primary monitoring questions for landscape dynamics include: 1) what is the spatial extent of land-cover types within and adjacent to the parks and the protected AT corridor, and 2) how have they changed over time? Through this study we wanted to test the hypotheses of 1) National

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Table 1
Selected NETN parks and AT segments for the study.

Site name	Abbreviation	State	Size
Acadia National Park	ACAD	Maine	19,222 ha
Marsh–Billings–Rockefeller NHP	MABI	Vermont	223 ha
Saint-Gaudens NHS	SAGA	New Hampshire	60 ha
Minute Man NHP	MIMA	Massachusetts	304 ha
Morristown NHP	MORR	New Jersey	682 ha
Saratoga NHP	SARA	New York	1036 ha
Roosevelt–Vanderbilt NHS	ROVA	New York	276 ha
Weir Farm NHS	WEFA	Connecticut	30 ha
Selected AT segments	APPA		
1. Whitecap Mountain		Maine	7 km
2. Saddleback Mountain		Maine	21 km
3. Chateauguay–No-town Area		Vermont	33 km
4. Hanover		New Hampshire	21 km
5. Tyringham Valley and Sheffield		Massachusetts	48 km
6. Walkill Valley		New Jersey	38 km
7. Dunnfield Creek		New Jersey	46 km
8. Hawk Mountain Sanctuary		Pennsylvania	64 km
9. Rausch Gap/St. Anthony's wilderness		Pennsylvania	57 km
10. Cumberland Valley		Pennsylvania	27 km

park units of the NETN that close to urban areas or within historical suburban settings experienced more development adjacent to the park boundaries in the past three decades; and 2) Selected AT segments in the northeastern U.S. experienced significant amount of land-cover change along the central path of the trail.

Remote sensing is a proven technology that is effective for mapping and characterizing cultural and natural resources (*e.g.*, Jensen, 1996; Campbell, 1997; Welch et al., 2002). Remote sensing allows observation and measurement of biophysical characteristics of the landscape, and tracking of changes in landscapes over time (Parmenter et al., 2003; Wang & Moskovits, 2001). Remote sensing change detection can be used to discern and simulate areas that have been altered by natural or anthropogenic processes (Jantz et al., 2003; Hansen et al., 2002). Change detection has often been discussed in the literature (Wilkinson et al., 2008; Rhemtulla et al., 2007; Woodcock & Ozdogan, 2004; Walker, 2003; Rogan et al., 2002; Hayes & Sader, 2001; Mas, 1999; Roberts et al., 1998; Lambin & Strahler, 1993; Mouat et al., 1993). Coppin et al. (2004) conducted a thorough review on change detection methods in ecosystem monitoring. Kennedy et al. (2009-this issue) has reviewed extensively the application issues of change detection in management of protected areas.

Understanding the magnitude and pattern of land-cover change helps establish a landscape context for the parks and protected areas, and offers resource managers a better understanding of how park ecosystems fit into the broader landscape. The intent of this study was to provide essential baseline data about the general land-cover types and landscape context in the vicinity of the NETN park units in the past three decades and to demonstrate an implementation of a protocol for revealing the past changes and monitoring future changes. The study had three primary objectives:

- Document general land-cover types within and surrounding selected NETN National Parks for 3 time periods: the mid-1970's, late-1980's, and 2002.
- Quantify changes in land-cover types within and adjacent to NETN parks using selected buffer areas for the three time periods.
- Reveal and assess patterns of land-cover change on the NETN parks and neighboring protected lands.

This study focused on eight (8) NETN park units, including National Park, National Historical Park (NHP) and National Historical Site (NHS), and ten (10) segments along the AT from Maine to Pennsylvania. The AT segments, totaling 362 km, were selected based on observed changes and on the potential for future change as

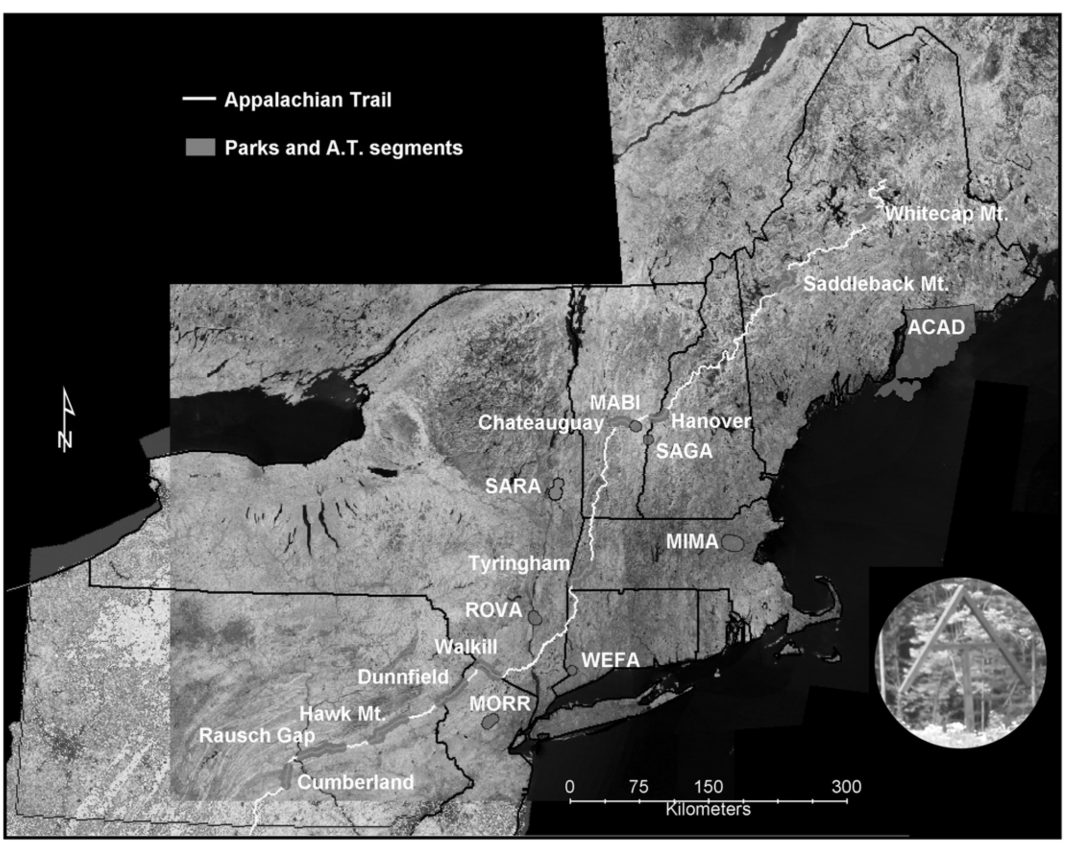

Fig. 1. Locations of selected NETN park units and AT segments displayed on top of a mosaic of Landsat images.

perceived by resource managers (Table 1, Fig. 1). Most of the NETN parks were established to preserve and protect significant cultural resources. Acadia (ACAD) is the only National Park in the NETN and hosts a diverse array of cultural, natural, and geologic resources. The AT extends along the entire Appalachian Mountain range in the eastern United States. The trail corridor is a minimum of 305 m wide and spans 3500 km from Maine to Georgia. The Trail repeatedly traverses the major elevation, latitudinal, ecological and cultural gradients that characterize the eastern United States. The AT and its protected corridor is an ideal transect to gauge changes in the environment caused by urbanization, recreational use, acid precipitation, exotic species, and climate change. Although the roughly 300 m-wide AT central corridor is protected, the adjacent landscape has been changing in the past decades. A recent study analyzed the extent and spatial distribution of forest clearing along a 16 km-wide corridor centered on the AT, and concluded that managed forest harvests in northern New England accounted for 76.8% of forest clearing within the corridor (Potere et al., 2007). This result highlights the importance of documenting baseline conditions and monitoring change of land-cover types along the AT corridor.

2. Methods

A variety of remote sensing change detection methodologies have been developed and evaluated over the past twenty years (Rogan et al., 2002; Woodcock & Ozdogan, 2004; Healey et al., 2005). The purpose of this paper is to provide a case study using existing remote sensing change detection methods that benefits park resource management and monitoring.

We used Landsat remote sensing data as the primary data source for derivation of generalized land-cover information. Landsat satellites provide multispectral data from the early 1970s to the present. Given that the purpose of this project was to provide a general landscape characterization and change analysis instead of detailed vegetation and resource mapping, the spatial resolution of Landsat data was appropriate. Data availability and cost were also a consideration. At the time this project was initiated a significant amount of Landsat data was available at no cost from on-line open resources such as the Global Land Cover Facility (GLCF) (http://glcf.umiacs.umd.edu/index.shtml) or at low cost from other data archives. As the intent of this study was to reveal the general trends of land-cover change and landscape context, the difference in spatial resolution between MSS and TM/ETM+ images was not a concern as long as we obtained areas of land-cover types. We searched for Landsat data that represented the best match in time frame and, if possible, were close to the anniversary of image acquisition in order to reduce seasonal effects. We ultimately acquired and processed thirty-three (33) scenes of Landsat images, eleven (11) scenes for each of the three time periods: the early and middle of 1970s (MSS data), late 1980s (TM data), and 2002 (ETM+ data). We projected all images into Universal Transverse Mercator (UTM) map coordinates and, when necessary, conducted geometric rectification with orthorectified Landsat ETM+ images as the base.

Instead of processing entire scenes of Landsat images we applied a 5-km buffer on park boundaries and the AT central line to subset Landsat images for land-cover classifications of the study sites. The only exception to this procedure was Acadia National Park. ACAD contains islands and separated units either completely or partially surrounded by ocean and bay waters. We therefore chose to subset a larger section from the Landsat images that covered all segments of ACAD for the purpose of land-cover classification (Fig. 2a).

We defined a generalized classification scheme that included nine land-cover categories: *Urban*, *Herbaceous Vegetation*, *Deciduous Forest*, *Coniferous Forest*, *Mixed Forest*, *Water*, *Wetlands*, *Barren Lands*, and *Bare Rocks*. We added an additional category (*Regrowth Forest*) for the Whitecap Mountain and Saddleback Mountain segments of the AT to reflect significant logging in the past and subsequent regrowth of forest adjacent to these two segments of the AT.

We conducted ground observation and verification under the guidance of NPS and AT scientists, land managers and volunteers. Field

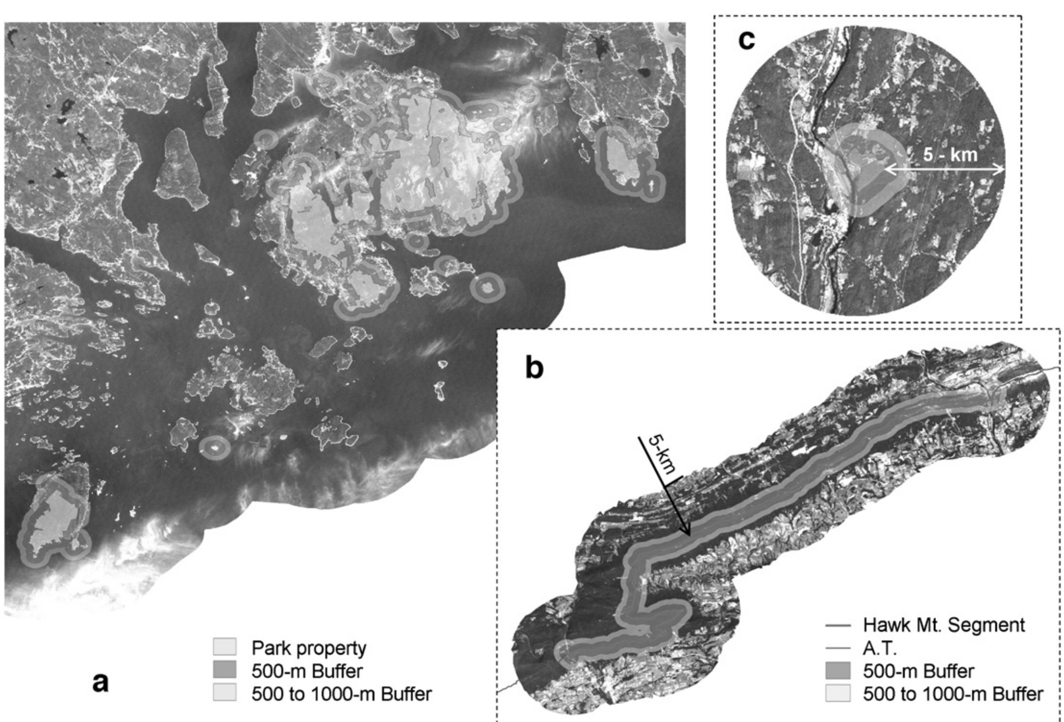

Fig. 2. Examples of subset Landsat images and the buffers for a. Acadia National Park, b. Saratoga NHP, and c. the AT Hawk Mountain segment. The subset image for ACAD site covers a larger area than the 5-km buffer zone.

observations provided an essential, independent reference for verifying land-cover types within the Landsat scenes as well as enhanced field knowledge for image interpretation and accuracy assessment. Since the ground referencing data were intended for supporting three time periods of Landsat images, we paid special attention to the locations where the landscape had been altered and land use had changed over the past thirty years. With the guidance of land managers and volunteers, we identified sites with changed land use and marked them on the hard copy of Landsat images and airphotos. These locations were later used as reference sites for examining the classification results. We used Trimble® ProXR and GeoXT Global Positioning System (GPS) units to record locations of field transects and points of interest. We recorded the general characteristics of the landscape and associated information, as well as a set of georeferenced field photographs at points of interest using a Kodak® DC265 Field Imaging System. We collected about 2800 georeferenced digital photographs which, when augmented with GPS data, effectively identified locations and characteristics of the landscape within each study site. We combined the field photographs and GPS point data to create a virtual field reference database (Lunetta et al., 2001) for each studied park unit and AT segment within a geographic information system (GIS). Fieldwork confirmed the correspondence between spectral features on Landsat images and land-cover types and patterns on the ground, as well as the changes that had occurred in the past. The virtual field reference database and georeferenced field photos provide benchmark data for long term monitoring of landscape context of the study areas.

We employed supervised, unsupervised, and stratified classifications for obtaining the land-cover data. We selected multiple signatures to represent the spectral variations for each of the land-cover types. We then cross checked with USGS National Land Cover Dataset (NLCD) of 1992 and 2001 (Homer et al., 2000), National Wetland Inventory (NWI) data, and NPS Vegetation Mapping Project data for major discrepancies between classification results and those reference datasets. For example if unlikely urban land existed from supervised classifications that were clearly in contradictory with NLCD data on urban land areas, we reexamined those pixel areas and modified the signatures accordingly to improve supervised classifications. We checked the classifications of late 1980s and 2002 Landsat images against the NLCD 1992 and NLCD 2001, respectively, to get a closer time match. The same practice applied with the NWI data. Upon finishing the supervised classification, we recoded the classes into appropriate land-cover categories, which resulted in final land-cover types defined by our classification scheme.

The NPS Vegetation Mapping Program is a cooperative effort by the USGS and NPS to classify, describe, and map vegetation of more than 270 national park units across the United States (http://biology.usgs.gov/npsveg/). The NPS vegetation mapping project utilizes the national vegetation classification standard as the classification scheme. The maps consisted of vector GIS data that defined boundaries of vegetation types based on plot information and manual delineations from aerial photos. The standard minimum mapping unit was 0.5 ha and the thematic accuracy was >80% per class. ACAD, Roosevelt–Vanderbilt NHS (ROVA), and Saratoga NHP (SARA) were the only park units in this study that had available NPS vegetation mapping project data at the time of image classifications. For those three sites we employed stratified classification technique in the land-cover mapping so that the vegetation mapping data can be referenced.

Stratification involves segmentation of an image into focused areas and categories based on existing GIS land-cover data in order to improve a classification (Wang et al., 2007). We used NPS vegetation mapping data as the base line to mask corresponding pixels for each vegetation/land-cover category from the subset of Landsat images, so that the follow up classifications were focused on each of the vegetation/land-cover types identified by the vegetation mapping projects. We began by rasterizing the vector GIS format vegetation data from ACAD, SARA and ROVA, matching the pixel size with the Landsat data. We then ran image-to-image geometric rectification on the rasterized vegetation map using the georeferenced Landsat-7 ETM+ images as a base. Next, we separated Landsat data into segmented images for further classification by using vegetation types and other land-cover categories in the rasterized NPS vegetation map data as the mask to extract pixels from Landsat images. We then ran unsupervised classifications on each of the segmented images into 20 spectral clusters and labeled the spectral clusters upon finishing the classification. The dominant category for the spectral clusters should be the vegetation or land-cover type defined by the masking type from the NPS vegetation map data. Pixels that had distinct spectral differences from the masked type were labeled into corresponding types defined by the classification scheme. We repeated this process until all the segmented images were classified and correctly labeled. We then mosaicked all classified segmented images to create the final land-cover map and conducted visual comparisons to assess the agreement between the NPS vegetation mapping project data and the classification results. This allowed us to identify and reduce conflicts between data products as well as use the NPS vegetation mapping data to monitor landscape context.

We employed equalized random sampling in our accuracy assessment by selecting 50 reference pixels for each land-cover category for each study site and for each time period. We interpreted the reference pixel samples directly from the Landsat imagery, using our understanding of the spectral features of land-cover types with field GPS photos as the guide, as well as referencing the available aerial photographs and GIS data products. We use error matrices to report the agreement between classification results and reference samples.

We adopted the post-classification comparison method to obtain changes in areas of land-cover type. Post-classification comparison, also known as delta classification, involves independently produced spectral classification data from each end of the time interval of interest, followed by comparison of data to detect changes in cover type (Coppin & Bauer, 1996; Mas, 1999; Coppin et al., 2004). The principal advantage of post-classification comparison is that the images are separately classified, thereby minimizing the problem of radiometric calibration between dates (Song et al., 2001) and reducing the amount of data pre-processing. By choosing an appropriate classification scheme, post-classification comparison can also be made insensitive to a variety of transient changes. With appropriately developed land-cover maps in separate time periods, the class changes during the time interval and transition rate between classes can be calculated (Hall et al., 1991). However, because post-classification change analyses depend on separately classified land-cover data for extracting the change analysis, error propagation is a major concern. The final accuracy of a change analysis is very close to the multiplied accuracy of each individual classification, and accumulated errors may mislead the interpretation of the change analysis. Given that post-classification change detection is an easy-to-execute and easy-to-communicate method to the land managers, we decided to use this method for this study.

We applied spatial buffers to extract land cover and landscape context information in different buffer zones. Our 500-m and 1-km buffer sizes roughly corresponded to distances where ecological edge effects have been noted for birds (approximately 500 m) and mammals (up to 900 m), and our 5-km buffer begins to approximate the land area needed to support large carnivores (up to 200,000 ha) (Kennedy et al., 2003). These buffer widths are similar to buffer sizes used by other authors (e.g., Brazner et al., 2007), and we chose these buffer sizes because they were likely to be ecologically relevant and because they allowed us to explore how land-cover changes with increasing distance from parks. The appropriate buffer sizes for a given analysis should be carefully chosen to match the purpose of the analysis, and may vary from the sizes that we chose. The buffering analysis provided four groups of land cover and landscape context information including: 1) within the park boundary (not applicable for AT segments); 2) the zone within

500 m of the park boundary/AT central line; 3) the zone within 1 km of the park boundary/AT central line; and 4) the zone within 5 km of the park boundary/AT central line (Fig. 2).

3. Results

Comparisons of land-cover data provide information on the spatial distribution and magnitude of land-cover changes within park boundaries and in adjacent buffer zones. Because changes in urban and forest lands represent the main land-cover changes in the study areas, we summarize changes in these two categories for the parks and AT segments as examples of our analysis. Overall, the parks showed a large increase in urban area between the 1970s and 2002 within the defined buffers (172% within 500 m, 181% within 1 km, and 212% within 5 km) and a decrease in forested area (17 to 18% in each buffer) (Fig. 3). The AT segments showed a similar increase in urban area (169% within 500 m and 189% within 1 km), but lower losses of forested area (about 2% in each buffer zone) (Fig. 4).

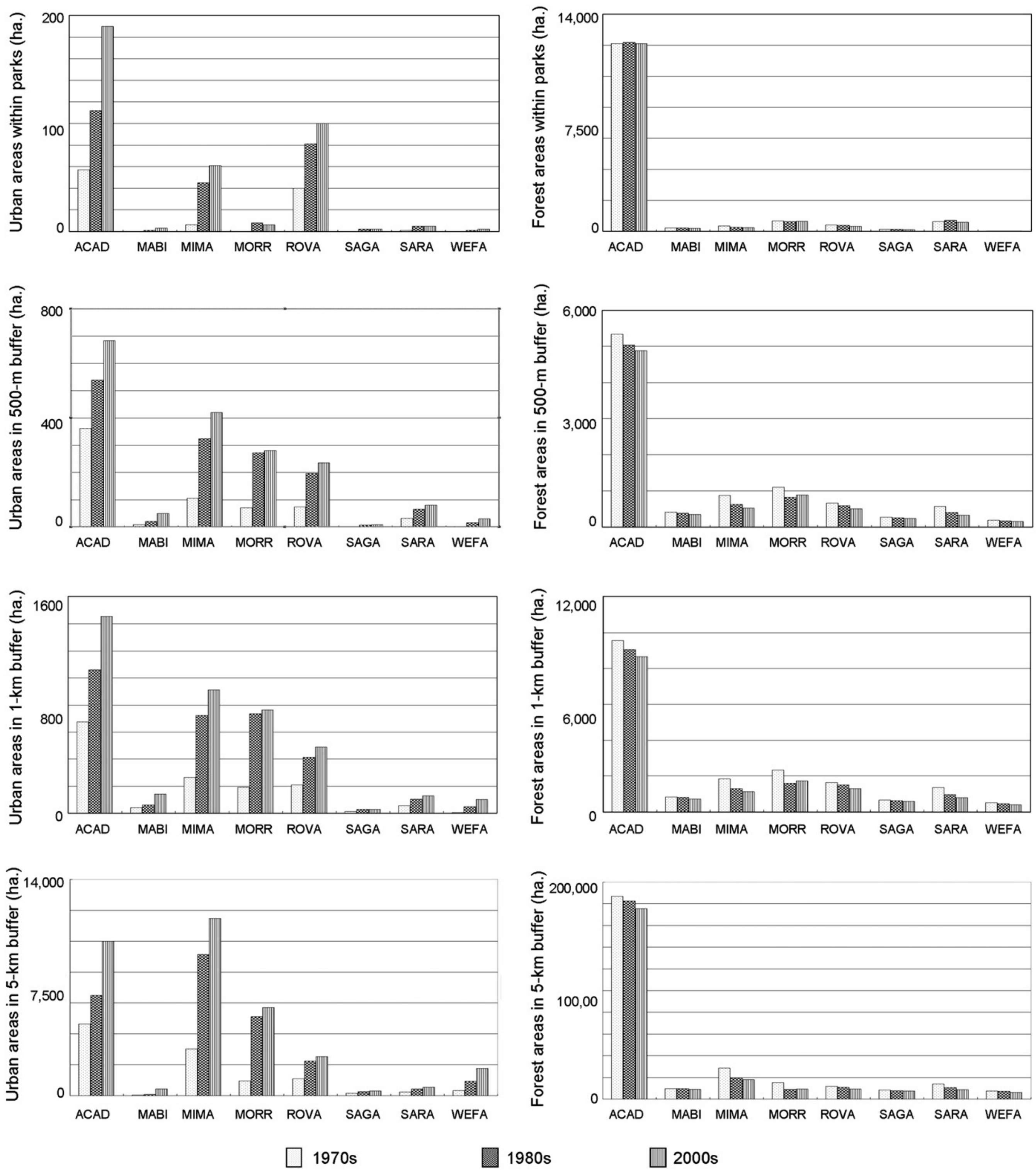

Fig. 3. A comparison of urban and forest lands within park boundaries and adjacent buffer zones in three time periods.

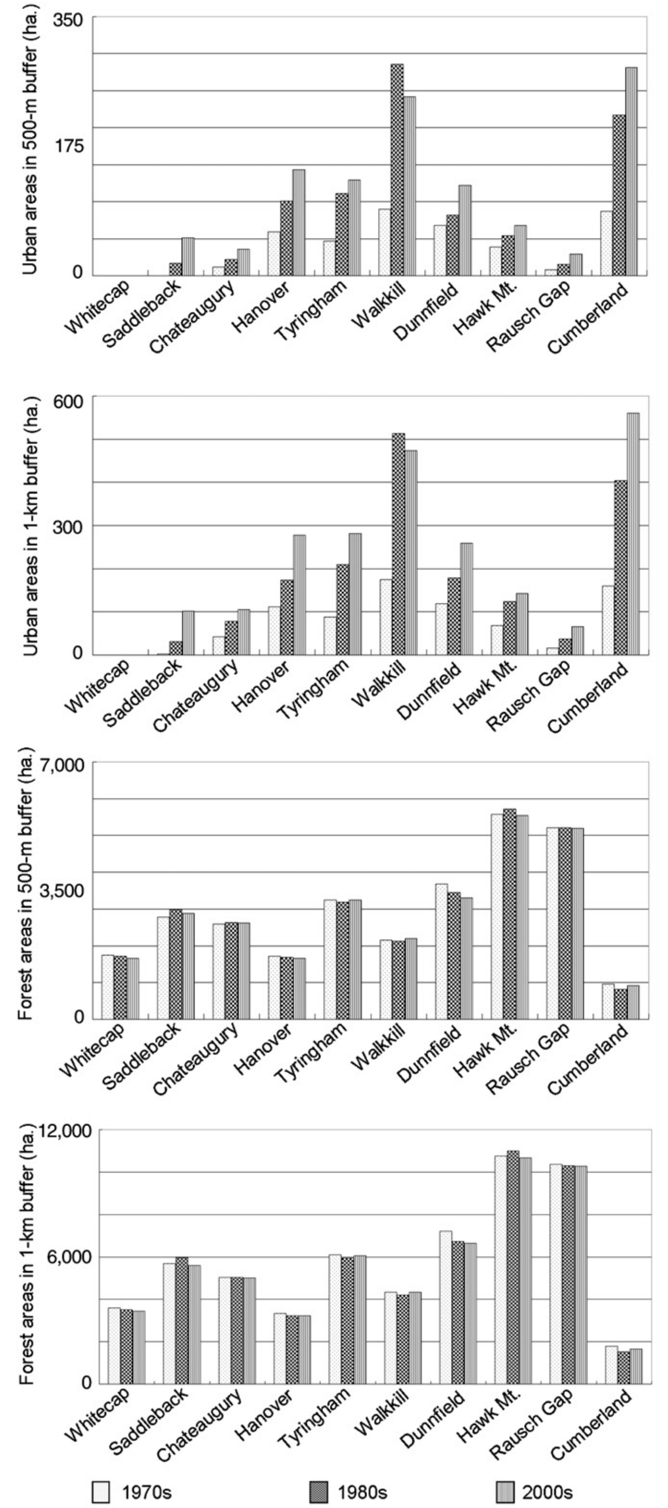

Fig. 4. A comparison of urban and forest lands adjacent to the central line of selected AT segments in three time periods.

Acadia, the only large national park in the NETN, protects the highest rocky headlands on the Atlantic shore of the United States. Urban development is an important concern for resource management at the park. Within the ACAD boundary, urban land changed from 57 ha to 190 ha, a 233% increase, between 1976 and 2002. Urban land approximately doubled in each of the buffer zones over the same period. Development along the gateway to the park (e.g., Route 3) and urban centers (e.g., Bar Harbor, Northeast Harbor, and Southwest Harbor) contribute to the changes of land cover and landscape context. The total areas of deciduous, coniferous and mixed forests within the ACAD boundary were stable between 1976 and 2002. However, within the 500-m buffer zone forest declined 5.6% between 1976 and 1986, and 3.1% between 1986 and 2002. The same pattern of five to 10% decline in forest area was seen within the 1-km and 5-km buffer zones.

Comparing maps of the main section of ACAD based on the NPS vegetation mapping project data (Fig. 5a), the USGS NLCD 2001 data (Fig. 5b) and the 2002 land-cover data from this study (Fig. 5c) illustrates some important differences. The NPS vegetation map represents land-cover types in manually delineated polygons. The NLCD data, as a national level dataset, may not reflect the variation of localized land covers within the specific park area well. For example, the covers of coniferous forest are overwhelmed in the NLCD map for the ACAD as shown in Fig. 5b. The map from our stratified classifications represents the variation of land-cover types of the park. Since our subset areas are more focused and we had more field observations to support our training signature selection, the final land-cover data, after cross checking with NLCD and other reference datasets, should reflect land-cover variation more closely than simply adopting land-cover categories from the NLCD or NPS vegetation mapping project.

The summarized overall classification accuracies for each study site and time period provide information regarding the classification results (Table 2). The assessment of classification accuracy for Acadia National Park serves as an example of our evaluation (Table 3). The user's and producer's accuracies, as well as the overall accuracy, report the contribution of errors from the classification of each land-cover type. The barren land/bare rocks and herbaceous vegetation achieved lower accuracies than other categories. The category of bare rocks was applied to Acadia National Park and the Whitecap Mountain and Saddleback Mountain segments of the AT to reflect the stony coastal shoreline and granite summits such as the Cadillac Mountain.

4. Discussion and conclusion

Monitoring landscape dynamics is important because changes within and adjacent to parks and protected areas can alter water quality and flow regimes, increase the likelihood of invasive plant and animal range expansions, reduce contiguous forest, and influence ambient sounds and clear night skies, among other impacts (Mitchell et al., 2006; Theobald, 2001). Monitoring land-cover changes will guide decision-making for resource management of these protected lands. This study produced land-cover maps across a large region, and provides the NETN with approximately 30 years of baseline data for understanding land-cover changes and landscape context in and around the parks and protected linear corridor of the AT. The baseline data will be used for years to come, and will provide a foundation for monitoring future changes in land cover.

The recent National Resources Inventory (NRI: http://www.nrcs.usda.gov/) report singled out that the impact of development on rural nonfederal land is a concern in the balancing of development needs with conservation of natural resources. Land conversions for developed uses can result in fragmentation of landscape, leading to diminished values for wildlife, water management, open space, and aesthetic purposes, among others (NRI, 2003). The buffer and change analyses of this study concluded that all of the studied park units and almost all of the selected AT segments experienced significant increases of urban land cover in immediately adjacent areas and extended buffer zones. The NRI reports indicate that the increase of developed land for the states of Maine, New Hampshire, Vermont, Massachusetts, Connecticut, New York, New Jersey and Pennsylvania, are far less than the increasing rates of urban lands adjacent to the park units and AT segments that are situated in the states.

In Maine, for example, statewide developed land increased from 2% of the area of the state in 1982, to 3% in 1987, to 3% in 1992 and to

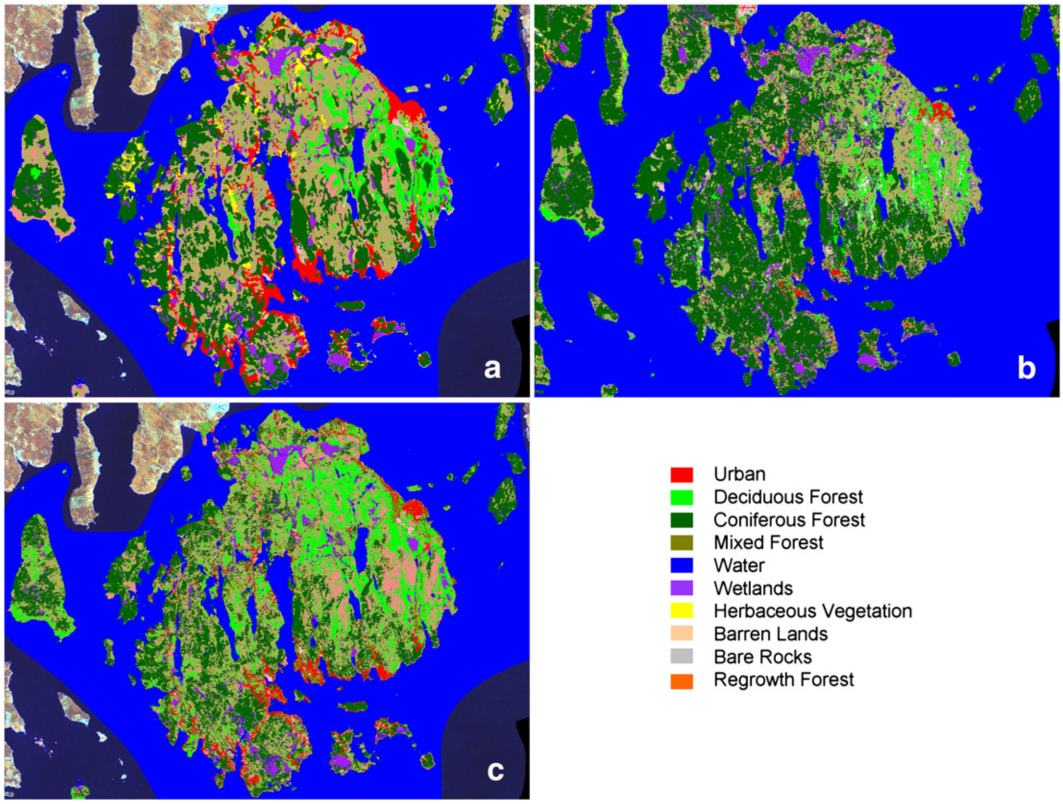

Fig. 5. A comparison of NPS vegetation mapping project data (a), NLCD 2001 data (b) and the 2002 land-cover data for selected portion of the Acadia National Park generated from the stratified classification (c).

4% in 1997. The forested land changed from 85% of the area of the state in 1982, to 89% in 1987, to 90% in 1992 and to 89% in 1997. The changes of urban and forested land within and adjacent the Acadia National Park, however, represent a different development pattern. Along the AT, the Whitecap Mountain segment in Maine is the only segment in the study sites that had no sign of urban development adjacent to the trail. The changes of forested land in the Whitecap Mountain and Saddleback Mountain segments reflect the effect of managed forest on the landscape that agreed with the pattern revealed by the NRI data.

In Vermont, the NRI data show that developed land increased from 4% of the area of the state in 1982, to 5% in 1987, and remained to 5% in 1992 and 1997, respectively. The forested land changed from 68% of the area of the state in 1982, to 71% in 1987, and remained as 71% in 1992 and 1997. At Marsh–Billings–Rockefeller NHP in Vermont, reforestation has occurred in many areas from agricultural abandonment, resulting in naturally regenerated northern hardwood forests. Forest cover within the park boundary did not change much over the years, but declined in the buffer areas. Urban land did not change within the MABI boundary, but it tripled between 1978 and 1989 and then doubled between 1989 and 2002 within the 0.5 km buffer. For the Chateauguay segment of the AT situated in central Vermont, urban land increased within defined buffer zones and forested land didn't change much, which showed the same pattern as NRI data indicated. The AT segments in New Hampshire and Massachusetts experienced significant urban development within the immediately adjacent buffer zone of the central path. Although the Walkill Valley and Dunnfield Creek segments in New Jersey are protected by public land acquisition efforts, urban increases along the AT were evident.

The NRI data indicate that for the state of Pennsylvania, while developed land increased from 10% of the area of the state in 1982, to 11% in 1987, to 12% in 1992 and to 14% in 1997, forested land accounted for 54% in 1982, 1987 and 1997 and 53% in 1992. For the AT in Pennsylvania, forested land in the Hawk Mountain segment has remained stable while urban area has doubled within the buffers surrounding this segment. The Rausch Gap/St. Anthony's Wilderness segment is the largest undeveloped roadless section of the AT in the mid-Atlantic region. Although not a federally designated wilderness, it is a significant tract of unbroken public land in central Pennsylvania. Urban land doubled in each of the time periods between 1973–1989 and 1989–2002, while forested land declined slightly in both the 0.5 and 1 km buffers. The Cumberland Valley segment is a rapidly developing residential and commercial area, in part due to major transportation corridors that traverse the area. Historically the valley has been a significant agricultural area. The center of the valley now is home to numerous trucking terminals served by the major east/west

Table 2
Overall classification accuracy and kappa coefficient for each site, and year of data acquisition and type of data.

Site name	The year, overall accuracy and kappa coefficient		
	1970s (MSS)	1980s (TM)	2000s (ETM+)
ACAD	1976, 84.8%, 0.83	1986, 86.5%, 0.85	2002, 87.0%, 0.85
MABI	1978, 89.0%, 0.87	1989, 85.7%, 0.83	2002, 88.6%, 0.87
SAGA	1978, 90.3%, 0.87	1989, 88.3%, 0.86	2002, 85.5%, 0.83
MIMA	1974, 85.1%, 0.83	1987, 91.4%, 0.90	2002, 87.0%, 0.85
MORR	1976, 87.1%, 0.85	1988, 84.6%, 0.82	2002, 89.4%, 0.88
SARA	1976, 84.8%, 0.83	1989, 88.29%, 0.86	2002, 85.7%, 0.83
ROVA	1973, 91.4%, 0.90	1988, 92.0%, 0.91	2002, 90.1%, 0.89
WEFA	1973, 87.1%, 0.85	1989, 85.1%, 0.83	2002, 86.2%, 0.84
APPA segments			
11. Whitecap	1976, 86.3%, 0.84	1987, 92.57%, 0.91	2002, 90.6%, 0.89
12. Saddleback	1972, 81.7%, 0.78	1986, 89.5%, 0.88	2002, 88.8%, 0.87
13. Chateauguay	1976, 68.3%, 0.64	1987, 70.3%, 0.66	2002, 78.3%, 0.75
14. Hanover	1978, 82.0%, 0.79	1989, 83.3%, 0.81	2002, 80.0%, 0.77
15. Tyringham	1973, 87.4%, 0.85	1989, 88.3%, 0.86	2002, 86.9%, 0.85
16. Walkill Valley	1975, 70.0%, 0.65	1988, 83.1%, 0.80	2002, 80.0%, 0.77
17. Dunnfield Creek	1975, 76.0%, 0.72	1988, 81.4%, 0.78	2002, 81.43%, 0.78
18. Hawk Mountain	1973, 72.8%, 0.68	1989, 83.4%, 0.80	2002, 79.7%, 0.76
19. Rausch Gap	1973, 79.0%, 0.79	1989, 86.7%, 0.84	2002, 90.3%, 0.88
20. Cumberland	1973, 82.3%, 0.79	1989, 83.3%, 0.80	2002, 83.0%, 0.80

Table 3
An example of accuracy assessment for the land-cover data developed from classification of Landsat data for the Acadia National Park in 2002, 1986 and 1976.

ACAD		Reference data									
		U	DF	CF	MF	W	WL	HV	BR	Totals	User's accuracy
2002 land-cover data	U	**42**	1	1	1	0	0	0	2	50	84.00%
	DF	0	**45**	1	2	0	0	1	1	50	90.00%
	CF	4	1	**40**	4	0	0	0	1	50	80.00%
	MF	3	2	0	**43**	0	1	1	0	50	86.00%
	W	0	0	0	0	**49**	0	0	1	50	98.00%
	WL	1	0	0	1	0	**47**	1	0	50	94.00%
	HV	5	1	0	1	0	0	**43**	0	50	86.00%
	BR	2	2	0	0	0	1	2	**39**	50	78.00%
	Totals	57	52	42	52	49	49	48	44	400	
	Producer's accuracy	73.68%	86.54%	95.24%	82.69%	100%	95.92%	89.58%	88.64%		Overall 87.00%
1986 land-cover data	U	**50**	0	0	0	0	0	0	0	50	100.00%
	DF	1	**44**	0	2	0	1	2	0	50	88.00%
	CF	0	1	**45**	2	0	2	0	0	50	90.00%
	MF	2	0	4	**43**	0	1	0	0	50	86.00%
	W	0	0	0	0	**49**	0	0	1	50	98.00%
	WL	0	0	0	0	0	**50**	0	0	50	100.00%
	HV	5	3	1	2	0	0	**31**	8	50	62.00%
	BR	3	1	1	3	0	7	1	**34**	50	68.00%
	Totals	**61**	**49**	**51**	**52**	**49**	**61**	**34**	**43**	400	
	Producer's accuracy	81.97%	89.80%	88.24%	82.69%	100%	81.97%	91.18%	79.07%		Overall 86.50%
1976 land-cover data	U	**50**	0	0	0	0	0	0	0	50	100.00%
	DF	0	**44**	1	1	0	1	1	2	50	88.00%
	CF	1	1	**42**	4	0	1	1	0	50	84.00%
	MF	1	2	2	**44**	0	0	0	1	50	88.00%
	W	0	0	0	0	**50**	0	0	0	50	100.00%
	WL	0	0	0	0	0	**47**	0	3	50	94.00%
	HV	8	5	0	1	0	1	**26**	9	50	52.00%
	BR	7	1	1	0	0	0	5	**36**	50	72.00%
	Totals	67	53	46	50	50	50	33	51	400	
	Producer's accuracy	74.63%	83.02%	91.30%	88.00%	100.00%	94.00%	78.79%	70.59%		Overall 84.75%

and north/south routes. This is the main factor that caused land-cover change in this segment.

The temporal variation of land-cover change reveals the time period that major development and changes occurred. Changing land-use patterns around Morristown NHP in northern New Jersey have altered the character of the area from farmed or hardwood forested areas intersected by streams to low density residential development, expanded networks of roads, and commercial and recreational development. The NRI data show that for the state of New Jersey, developed land increased from 24% of area of the state in 1982, to 32% in 1987, to 33% in 1992 and to 38% in 1997. Although urban land did not change much within the park boundary, the increase was significant in all buffer zones, particularly in the time period between 1976 and 1988 when a major development occurred adjacent to the park. As the available land became saturated, the change in urban area slowed down between 1988 and 2002 within the buffer zones. The results answered the hypothesized questions that the national park units of the NETN that close to urban area or within historical suburban settings experienced more development adjacent to the park boundaries in the past three decades than the statewide development patterns. Selected sensitive AT segments in the north-eastern U.S. experienced significant amount of urban development along the central path of the trail.

This study confirms that the NETN park units are facing critical management challenges due to the complexity and variability of landscape contexts outside the boundaries of their protected areas. The NETN is not the only I&M network that faces such a challenge. Other studies show that rural areas in the American West, for example, are undergoing a dramatic transition in demography, economics, and ecosystems (Riebsame et al., 1997; Theobald, 2001; Beyers & Nelson, 2000). Studies in the Greater Yellow Stone Ecosystem suggested that as natural amenities attract people and commerce, the resulting land-use change threaten biodiversity of the protected areas and challenge efforts to sustain local communities and ecosystem (Hansen et al., 2002; Rasker & Hansen, 2000). Urban development adjacent the National Parks and protected areas may be shrinking the natural buffers of those reserves and altering the ecosystems within them. Management of park units and protected areas will have to consider the context of the landscape as well as past and future changes in order to achieve the goal of science based analysis, synthesis, and modeling.

Though the AT is but a thin ribbon when viewed from a regional perspective, threats to the environment of the Appalachian Trail — including urbanization, recreational impacts, acid precipitation, exotic species, and climate change represent threats to this one of the most biologically diverse units of the National Park System (Doufour & Cristfield, 2008). The study for selected AT segments helps understand the status of sensitive areas of the AT corridor, or a MEGA-Transect, in landscape configuration. It provides baseline data for possible further studies in landscape composition, diversity, distances from sources, edge-to-area ratios, and ecotonal features that may structure the plant and animal communities (Rosenberg et al., 1997; Schweiger et al., 2000). In cases where some development may be acceptable, management of AT can be incorporated into conservation of existing landscape configuration so that AT can function as a connection between larger protected areas. The protocol of this case study will be suitable for other linear protected areas such as river-based parks, roadways, habitat corridors, or trails across the country.

Landsat data and the derivatives of land-cover maps allow several units to be mapped at once and over time. As USGS plans to provide Landsat image products at no cost upon request for the entire U.S. archive (Woodcock et al., 2008), free imagery will help promote remote sensing applications among user groups. The consistency of methods across the network should help the NPS assess how parks are differentially affected by natural and anthropogenic processes.

Land-cover change analysis is among the necessary tools for assessing landscape-scale impacts to park units, provided that we identify independent variables or processes that can be related to the changes observed with the remote sensing data. For example, assessing the impact of urbanization on specific subject in a given park, such as invasive species, can be conducted by observing the change of spatial patterns of the subject, and using spatial statistics to correlate severity of the subject with the urbanization or fragmentation processes that can be estimated from the land-cover data. Identifying these independent variables requires multidisciplinary expertise, with ecologists, botanists, biologists, hydrologists, and others working together and with land-cover data to answer questions about threats and processes with information in land-cover change and landscape context.

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References

Beyers, W. B., & Nelson, P. B. (2000). Contemporary development forces in the nonmetropolitan West: New insights from rapidly growing communities. *Journal of Rural Studies*, *16*, 459−474.

Brazner, J. C., Danz, N. P., Trebitz, A. S., Niemi, G. J., Regal, R. R., Hollenhorst, T., et al. (2007). Responsiveness of Great Lakes wetland indicators to human disturbances at multiple spatial scales: A multi-assemblage assessment. *Journal of Great Lakes Research*, *33*(Special Issue 3), 42−66.

Campbell, J. B. (1997). *Introduction to remote sensing.* New York: Guilford Press.

Coppin, P. R., & Bauer, M. E. (1996). Change detection in forest ecosystems with remote sensing digital imagery. *Remote Sensing Reviews*, *13*, 207−234.

Coppin, P., Jonckheere, I., Nackaerts, K., Muys, B., & Lambin, E. (2004). Digital change detection methods in ecosystem monitoring: A review. *International Journal of Remote Sensing*, *25*(9), 1565−1596.

Doufour, C., & Cristfield, E. (2008). *The Appalachian Trail MEGA-Transect, Harpers Ferry, WV, Appalachian Trail Conservancy, 2008.*

Fancy, S. G., Gross, J. E., Carter, S. L. (2009). Monitoring the condition of natural resources in U.S. National Parks. *Environmental Monitoring and Assessment, 151*, 161–174.

Gross, J. E., Nemani, R. R., Turner, W., & Melton, F. (2006). Remote sensing for the national parks. *Park Science*, *24*, 30−36.

Hall, F. G., Botkin, D. B., Strebel, D. E., Woods, K. D., & Goetz, S. J. (1991). Large-scale patterns of forest succession as determined by remote sensing. *Ecology*, *72*, 628−640.

Hansen, A. J., & Rotella, J. J. (2002). Biophysical factors, land use, and species viability in and around nature reserves. *Conservation Biology*, *16*, 1−12.

Hansen, A. J., Rasker, R., Maxwell, B., Rotella, J. J., Wright, A., Langner, U., et al. (2002). Ecology and socioeconomics in the New West: A case study from Greater Yellowstone. *Bioscience*, *52*, 151−162.

Hayes, D. J., & Sader, S. A. (2001). Comparison of change-detection techniques for monitoring tropical forest clearing and vegetation regrowth in a time series. *Photogrammetric Engineering & Remote Sensing*, *67*(9), 1067−1075.

Healey, S. P., Cohen, W. B., Zhiqiang, Y., & Krankina, O. N. (2005). Comparison of Tasseled Cap-based Landsat data structures for use in forest disturbance detection. *Remote Sensing of Environment*, *97*, 301−310.

Homer, C., Huang, C., Yang, L., Wylie, B., & Coan, M. (2000). Development of a 2001 national landcover database for the United States. *Photogrammetric Engineering and Remote Sensing*, *70*(7), 829−840.

Jantz, C. A., Goetz, S. J., & Shelley, M. A. (2003). Using the SLEUTH urban growth model to simulate the land use impacts of policy scenarios in the Baltimore–Washington metropolitan region. *Environment and Planning (B)*, *31*(2), 251−271.

Jensen, J. R. (1996). *Introductory digital image processing: A remote sensing perspective.* Upper Saddle River, New Jersey: Prentice-Hall.

Kennedy, R. E., Townsend, P. A., Gross, J. E., Cohen, W. B., Bolstad, P., Wang, Y. Q., et al. (2009). Remote sensing change detection tools for natural resource managers: Understanding concepts and tradeoffs in the design of landscape monitoring projects. *Remote Sensing Environment*, *113*, 1382−1396 (this issue).

Kennedy, C., Wilkinson, J., & Balch, J. (2003). *Conservation thresholds for land use planners.*Washington, D.C.: Environmental Law Institute 55 pp.

Lambin, E. F., & Strahler, A. H. (1993). Change-vector analysis in multitemporal space: A tool to detect and categorize land-cover change processes using high-temporal resolution satellite data. *Remote Sensing Environment*, *48*, 231−244.

Lunetta, R. S., Iiames, J., Knight, J., Congalton, R. G., & Mace, T. H. (2001). An assessment of reference data variability using a "virtual field reference database. *Photogrammetric Engineering & Remote Sensing*, *63*(6), 707−715.

Mas, J. F. (1999). Monitoring land-cover changes: A comparison of change detection techniques. *International Journal of Remote Sensing*, *20*(1), 139−152.

Mitchell, B. R., Shriver, W. G., Dieffenbach, F., Moore, T., Faber-Langendoen, D., Tierney, G., et al. (2006). Northeast Temperate Network Vital Signs Monitoring Plan. *Technical Report NPS/NER/NRTR-2006/059. National Park Service, Northeast Temperate Network, Woodstock, Vermont*.

Mouat, D. A., Mahin, G. G., & Lancaster, J. (1993). Remote sensing techniques in the analysis of change detection. *Geocarto International*, *2*, 39−50.

National Resources Inventory (2003). 2001 Annual NRI: Urbanization and Development of Rural Land, July 2003, Natural Resources Conservation Service, USDA, Washington D.C.

Parmenter, A. P., Hansen, A., Kennedy, R., Cohen, W., Langner, U., Lawrence, R., et al. (2003). Land use and land cover change in the Greater Yellowstone Ecosystem: 1975–95. *Ecological Applications*, *13*, 687−703.

Potere, D., Woodcock, C., Schnelder, A., Ozdogan, M., & Baccini, A. (2007). Patterns in Forest Clearing Along the Appalachian Trail Corridor. *Photogrammetric Engineering & Remote Sensing*, *73*(7), 783−791.

Rasker, R., & Hansen, A. J. (2000). Natural amenities and population growth in the Greater Yellowstone region. *Human Ecology Review*, *7*, 30−40.

Rhemtulla, J. M., Mladenoff, D. J., & Clayton, M. K. (2007). *Regional land-cover conversion in the U.S. upper Midwest: Magnitude of change and limited recovery (1850–1935–1993)Landscape Ecology*, *Vol. 22*. (pp. 57−75) Supplement 1/December, 2007.

Riebsame, W. E., Gosnell, H., & Theobald, D. (Eds.). (1997). *Atlas of the New West* New York: W.W. Norton.

Roberts, D.A., Batista, G.T., Pereira, J.L.G., Waller, E.K., & Nelson, B.W. (1998). Change identification using multitemporal spectral mixture analysis: application in Eastern Amazonia, In Remote sensing change detection: environmental monitoring methods and applications, Lunetta and Elvidge (Editors), Ann Arbor Press, Chelsea, Michigan, pp. 137–161.

Rogan, J., Franklin, J., & Roberts, D. A. (2002). A comparison of methods for monitoring multitemporal vegetation change using Thematic Mapper imagery. *Remote Sensing of Environment*, *80*, 143−156.

Rosenberg, D. K., Noon, B. R., & Meslow, E. C. (1997). Biological corridors: Form, function, and efficacy. *Bioscience*, *47*, 677−687.

Sala, O. E., Chapin, F. S., III, Armesto, J. J., Berlow, E., Blomfield, J., Dirzo, R., et al. (2000). Global biodiversity scenarios for the year 2100. *Science*, *287*, 1770−1774.

Schweiger, E. W., Diffendorfer, J. E., Holt, R. D., Pierotti, R., & Gaines, M. S. (2000). The interaction of habitat fragmentation, plant, and small mammal succession in an old field. *Ecological Monographs*, *70*, 383−400.

Shriver, G., Maniero, T., Schwarzkopf, K., Lambert, D., Dieffenbach, F., Owen, D., et al. (2005). *Appalachian Trail Vital Signs, National Park Service, Northeast Region, Boston, Massachusetts, November 2005*.

Song, C., Woodcock, C. E., Seto, K. C., Lenney, M. P., & Macomber, S. A. (2001). Classification and change detection using Landsat TM data: When and how to correct atmospheric effects? *Remote Sensing of Environment*, *75*, 230−244.

Soule, M. S. (1991). Conservation: Tactics for a constant crisis. *Science*, *253*, 744−750.

Theobald, D. M. (2001). Land use dynamics beyond the American urban fringe. *Geographical Review*, *91*, 544−564.

Walker, R. (2003). Mapping process to pattern in the landscape change of the Amazonian frontier. *Annuals of the Association of American Geographers*, *93*(2), 376−398.

Wang, Y., & Moskovits, D. K. (2001). Tracking fragmentation of natural communities and changes in land cover: Applications of Landsat data for conservation in an urban landscape (Chicago Wilderness). *Conservation Biology*, *15*(4), 835−843.

Wang, Y., Traber, M., Milstead, B., & Stevens, S. (2007). Terrestrial and submerged aquatic vegetation mapping in Fire Island National Seashore using high spatial resolution remote sensing data. *Marine Geodesy*, *30*(1), 77−95.

Welch, R., Madden, M., & Jordan, T. (2002). Photogrammetric and GIS techniques for the development of vegetation database of mountainous areas: Great Smoky Mountains National Park. *ISPRS Journal of Photogrammetry and Remote Sensing*, *55*, 53−68.

Wessels, K. J., De Fries, R. S., Dempewolf, J., Anderson, L. O., Hansen, A. J., Powell, S. L., et al. (2004). Mapping regional land cover with MODIS data for biological conservation: Examples from the Greater Yellowstone Ecosystem, USA and Pará State, Brazil. *Remote Sensing of Environment*, *92*, 67−83.

Wilkinson, D. W., Parker, R. C., & Evans, D. L. (2008). Change detection techniques for use in a statewide forest inventory program. *PE & RS*, *74*(7), 893−901.

Woodcock, C. E., Allen, R., Anderson, M., Belward, A., Bindschadler, R., Cohen, W., et al. (2008, May 23). *Free access to Landsat imageryScience*, *Vol. 320*. (pp. 1011).

Woodcock, C.E., & Ozdogan, M. (2004). Trends in land cover mapping and monitoring, Land Change Science (Gutman, editor), Springer, New York, pp. 367–377.