



Remote Sensing of Wetlands: Case Studies Comparing Practical Techniques

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ABSTRACT

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To plan for wetland protection and sensible coastal development, scientists and managers need to monitor the changes in coastal wetlands as the sea level continues to rise and the coastal population keeps expanding. Advances in sensor design and data analysis techniques are making remote sensing systems practical and attractive for monitoring natural and man-induced wetland changes. The objective of this paper is to review and compare wetland remote sensing techniques that are cost-effective and practical and to illustrate their use through two case studies. The results of the case studies show that analysis of satellite and aircraft imagery, combined with on-the-ground observations, allows researchers to effectively determine long-term trends and short-term changes of wetland vegetation and hydrology.

ADDITIONAL INDEX WORDS: *Wetland remote sensing, wetland case studies, remote sensor comparison, coastal ecosystems, sea level rise.*

INTRODUCTION AND BACKGROUND

Wetlands and estuaries are highly productive and act as critical habitats for a variety of plants, fish, shellfish, and other wildlife. Wetlands also provide flood protection, protection from storm and wave damage, water quality improvement through filtering of agricultural and industrial waste, and recharge of aquifers (Morris *et al.*, 2002; Odum, 1993). However, wetlands have been exposed to a range of stress-inducing alterations, including dredge and fill operations, hydrologic modifications, pollutant runoff, eutrophication, impoundments, and fragmentation by roads and ditches.

Recently, there has also been considerable concern regarding the impact of climate change on coastal wetlands, especially due to relative sea level rise, increasing temperatures, and changes in precipitation. Climate change is considered a cause for habitat destruction, shift in species composition, and habitat degradation in existing wetlands (Baldwin and Mendelsohn, 1998; Titus *et al.*, 2009). Coastal wetlands have already proved susceptible to climate change, with a net loss of 33,230 acres from 1998 to 2004 in the United States alone (Dahl, 2006). This loss was primarily due to conversion of coastal salt marsh to open saltwater. Rising sea levels not only can cause the drowning of salt marsh habitats but also can reduce germination periods (Noe and Zedler, 2001). The impact of global change in the form of accelerating sea level rise and

more frequent storms is of particular concern for coastal wetlands managers.

Vegetated wetlands are stable only when the marsh platform is able to accrete sediment at a rate equal to the prevailing rate of sea level rise. This ability to accrete is proportional to the biomass density of the plants, concentration of suspended sediment, time of submergence, and depth of the marsh surface and the tidal range. Many coastal wetlands, such as the tidal salt marshes along the Louisiana coast, are generally within fractions of a meter of sea level and will be lost, especially if the impact of sea level rise is amplified by coastal storms. Man-made modifications of wetland hydrology and extensive urban development will further limit the ability of wetlands to survive sea level rise. For instance, man-made channelization of the Mississippi River flow causes much of the river sediment to be carried into the Gulf of Mexico, rather than to be deposited in the wetlands along the Louisiana coast (Farris, 2005; Pinet, 2009).

County, state, and federal officials are concerned about the impact of climate change and sea level rise on fisheries, wetlands, estuaries, and shorelines; municipal infrastructure, such as water, wastewater, and street systems; storm water drainage and flooding; salinity intrusion into groundwater supplies; *etc.* (Nicholas Institute, 2010). To plan for wetland protection and sensible coastal development, scientists and managers need to monitor the changes in coastal ecosystems as the sea level continues to rise and the coastal population keeps expanding. Recent advances in sensor design and data analysis techniques are making some remote sensing systems practical and attractive for monitoring natural and man-induced coastal ecosystem changes. Hyperspectral imagers can differentiate

wetland types using spectral bands specially selected for a given application. High resolution multispectral mappers are available for mapping small patchy upstream wetlands. Thermal infrared scanners can map coastal water temperatures, while microwave radiometers can measure water salinity, soil moisture, and other hydrologic parameters. Synthetic Aperture Radars (SAR) help distinguish forested wetlands from upland forests. Airborne light detection and ranging (LIDAR) systems can be used to map wetland topography, produce beach profiles and bathymetric maps (Purkis and Klemas, 2011; Ramsey, 1995).

With the rapid development of new remote sensors, databases, and image analysis techniques, there is a need to help potential users choose remote sensors and data analysis methods that are most appropriate and practical for wetland studies (Phinn et al., 2000). The objective of this paper is to review and compare wetland remote sensing techniques that are cost-effective and practical and to illustrate their use through two case studies. The wetland sites and projects selected for the case studies are facing environmental problems, such as urban development in their watersheds or major vegetation and hydrologic changes due to rapid local sea level rise.

WETLAND AND LAND COVER MAPPING

For more than three decades, remote sensing techniques have been used successfully by academic researchers and government agencies to map and monitor wetlands (Dahl, 2006; Tiner, 1996). For instance, the U.S. Fish and Wildlife Service (FWS) has used remote sensing techniques to determine the biologic extent of wetlands for the past 30 years. Through its National Wetlands Inventory, FWS has provided federal and state agencies, the private sector, and citizens with scientific data on wetlands location, extent, status, and trends. To accomplish this important task, FWS has used multiple sources of aircraft and satellite imagery and on-the-ground observations (Tiner, 1996). Most states have also conducted a range of wetland inventories, using both aircraft and satellite imagery. The aircraft imagery frequently included natural color and color infrared images. The satellite data consisted of both high-resolution (1–4 m) and medium-resolution (10–30 m) multispectral imagery.

More recently, the availability of high spatial and spectral resolution satellite data has significantly improved the capacity for upstream wetland, salt marsh, and other coastal vegetation mapping (Jensen et al., 2007; Wang, Christiano, and Traber, 2010). Furthermore, new techniques have been developed for mapping wetlands and even identifying wetland types and plant species (Jensen et al., 2007; Klemas, 2009; Schmidt et al., 2004; Yang et al., 2009). Using hyperspectral imagery and narrow-band vegetation indices, researchers have been able to identify some wetland species and to make progress on estimating biochemical and biophysical parameters of wetland vegetation, such as water content, biomass, and leaf area index (Adam, Mutanga, and Rugege, 2010). Hyperspectral imagers may provide several hundred spectral bands; multispectral imagers use less than a dozen bands.

The integration of hyperspectral imagery and LIDAR-derived elevation has also significantly improved the accuracy

of mapping salt marsh vegetation. The hyperspectral images help distinguish high marsh from other salt marsh communities, using its high reflectance in the near-infrared region of the spectrum, and the LIDAR data help separate invasive *Phragmites australis* from low marsh plants (Yang and Artigas, 2010). Major plant species within a complex, heterogeneous tidal marsh have been classified using multitemporal, high-resolution QuickBird images, field reflectance spectra, and LIDAR height information. *Phragmites*, *Typha*, and *Spartina patens* were spectrally distinguishable at particular times of the year, likely due to differences in biomass and pigments and the rate at which these change throughout the growing season. Classification accuracies for *Phragmites* were high due to the uniquely high near-infrared reflectance and the height of this plant in the early fall (Gilmore et al., 2010).

High-resolution imagery is more sensitive to within-class spectral variance, making separation of spectrally mixed land cover types more difficult than when using medium-resolution imagery. Therefore, pixel-based techniques are sometimes replaced by object-based methods, which incorporate spatial neighborhood properties by segmenting or partitioning the image into a series of closed objects that coincide with the actual spatial pattern and then classifying the image. "Region growing" is among the most commonly used segmentation methods. This procedure starts with the generation of seed points over the whole scene, followed by grouping of neighboring pixels into an object under a specific homogeneity criterion. Thus, the object keeps growing until its spectral closeness metric exceeds a predefined break-off value (Kelly and Tuxen, 2009; Shan and Hussain, 2010; Wang, Sousa, and Gong, 2004).

Wetland health is strongly affected by runoff from land and its use within the same watershed. To study the impact of land runoff on estuarine and wetland ecosystems, a combination of models is frequently used, including watershed, hydrodynamic, water quality, and living resource models (Li et al., 2006; Linker et al., 1993). Most coastal watershed models require land cover or land use as an input. Knowing how the land cover is changing, these models, together with a few other inputs like slope and precipitation, can predict the amount and type of runoff into rivers, wetlands, and estuaries and how their ecosystems will be affected (Jensen, 2007). For instance, some models predict that severe degradation in stream water quality will occur when the agricultural land use in watersheds exceeds 50% or urban land use exceeds 20% (Tiner et al., 2000).

The Landsat Thematic Mapper (TM) has been a reliable source for land cover data (Lunetta and Balogh, 1999). Its 30-m resolution and spectral bands have proved adequate for observing land cover changes in large coastal watersheds (e.g., Chesapeake Bay). Figure 1 shows a land cover map of the Chesapeake Bay watershed derived from Landsat Enhanced Thematic Mapper Plus (ETM+) imagery. Thirteen land cover classes are mapped in Figure 1, including two wetland classes. Other satellites with medium-resolution imagers can also be used (Klemas, 2005).

As shown in Figure 1, the Chesapeake Bay watershed contains many streams and, consequently, upstream freshwater wetlands. Upstream wetlands are no less valuable than tidal marshes because they (1) improve the water quality of adjacent rivers by removing pollutants; (2) reduce velocity,

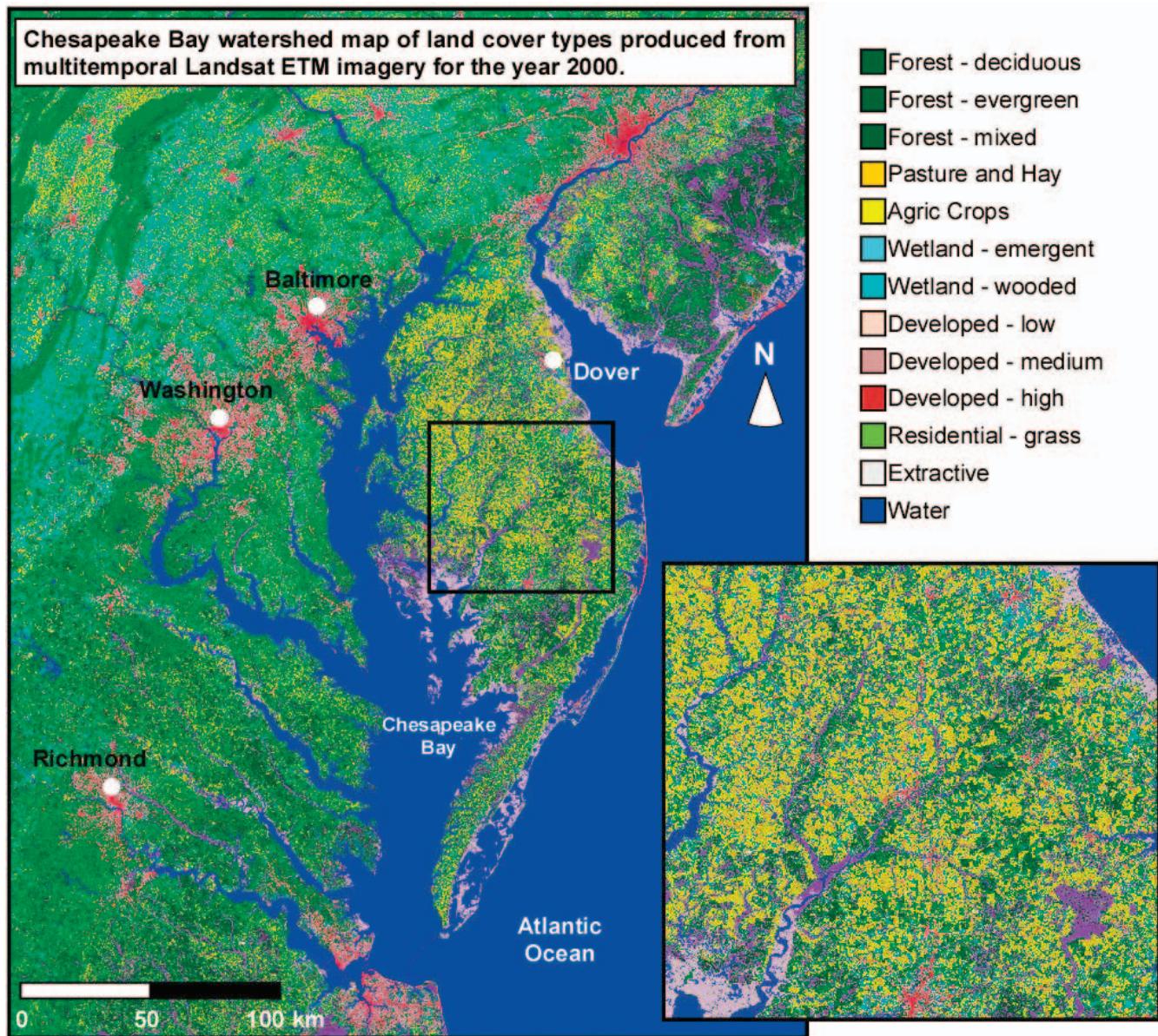


Figure 1. Chesapeake Bay watershed map of land cover types produced from multitemporal Landsat ETM+ imagery for 2000. (Modified with permission from Goetz *et al.*, 2004.)

erosion, and peak flow of floodwaters downstream; (3) provide habitat for wildlife; (4) serve as spawning and nursery grounds for many species of fish; and (5) contribute detritus to the aquatic food chain.

Originally, the Clean Water Act protected tidal marshes and freshwater wetlands. However, since the Supreme Court decisions in the SWANCC (2001) and Carabell/Rapanos (2006) cases, many isolated freshwater wetland are no longer protected by the Clean Water Act. State wetland managers are interested in how to find these wetlands, how to assess their ecologic integrity, and how to use this information to protect them and improve their condition or restore them (Tiner *et al.*,

2002). However, freshwater wetlands are small, patchy, and spectrally impure. Medium-resolution sensors, such as Landsat TM, miss some of these patchy wetlands and produce too many mixed pixels, increasing errors. Therefore, to map upstream, freshwater wetlands, managers need high spatial resolution and, in some cases, hyperspectral imagery. As a result, most upstream, freshwater wetlands are not mapped in Figure 1.

A typical digital image analysis approach for classifying coastal wetlands or land cover is shown in Figure 2. Before analysis, the multispectral imagery must be radiometrically and geometrically corrected. The radiometric correction reduces the influence of haze and other atmospheric scattering

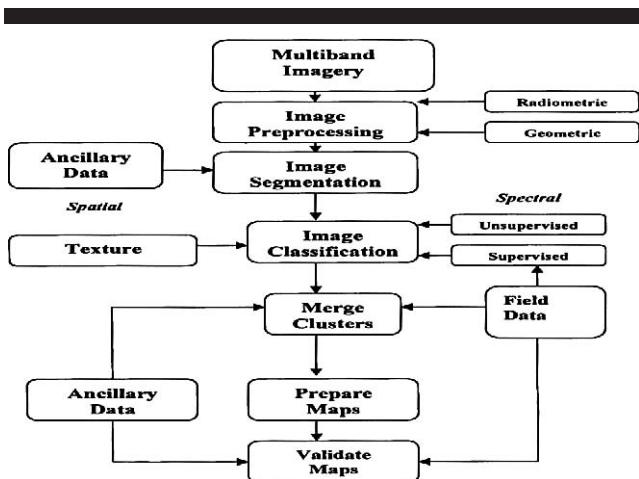


Figure 2. Typical image analysis approach.

particles and any sensor anomalies. The geometric correction compensates for the Earth's rotation and for variations in the position and attitude of the satellite. Image segmentation simplifies the analysis by first dividing the image into homogeneous patches or ecologically distinct areas. Supervised classification requires the analyst to select training samples from the data that represent the themes to be classified (Jensen, 1996). The training sites are geographic areas previously identified using field visits or other reference data, such as aerial photographs. The spectral reflectances of these training sites are then used to develop spectral "signatures," which are used to assign each pixel in the image to a thematic class.

Next, an unsupervised classification is performed to identify variations in the image not contained in the training sites. In unsupervised classification, the computer automatically identifies the spectral clusters representing all features on the ground. Training site spectral clusters and unsupervised spectral classes are then compared and analyzed using cluster analysis to develop an optimum set of spectral signatures. Final image classification is then performed to match the classified themes with the project requirements (Jensen, 1996). Throughout the process, ancillary data are used whenever available (e.g., aerial photos, maps, and field samples).

When studying small wetland sites, researchers can use aircraft or high-resolution satellite systems (Klemas, 2005). Airborne georeferenced digital cameras, providing color and color infrared digital imagery, are particularly suitable for accurate mapping or interpreting satellite data. Most digital cameras are capable of recording reflected visible to near-infrared light. A filter is placed over the lens that transmits only selected portions of the wavelength spectrum. For a single-camera operation, a filter is chosen that generates natural color (blue-green-red wavelengths) or color-infrared (green-red-near-infrared wavelengths) imagery. For a multiple-camera operation, filters that transmit narrower bands are chosen. For example, a four-camera system may be configured so that each camera filter passes a band matching a specific satellite imaging band, e.g., blue, green, red, and near-infrared bands

matching the bands of the IKONOS satellite multispectral sensor (Ellis and Dodd, 2000).

Digital camera imagery can be integrated with global positioning system position information and used as layers in a geographic information system for a range of modeling applications (Lyon and McCarthy, 1995). Small aircraft flown at low altitudes (e.g., 100–500 m) can be used to supplement field data. High-resolution imagery (0.6–4 m) can also be obtained from satellites, such as IKONOS and QuickBird (Table 1). However, cost becomes excessive if the site is larger than a few hundred square kilometers. In those cases, medium-resolution sensors, such as Landsat TM (30 m) and Satellite Pour l'Observation de la Terre (SPOT) (20 m), become more cost-effective.

Mapping submerged aquatic vegetation (SAV), coral reefs, and general bottom characteristics requires high-resolution (1–4 m) multispectral or hyperspectral imagery (Mishra *et al.*, 2006; Mumby and Edwards, 2002; Purkis *et al.*, 2002). Coral reef ecosystems usually exist in clear water and can be classified to show different forms of coral reef, dead coral, coral rubble, algal cover, sand, lagoons, different densities of sea grasses, *etc.* SAV sometimes grows in more turbid water and thus is more difficult to map. Aerial hyperspectral scanners and high-resolution multispectral satellite imagers, such as IKONOS and QuickBird, have been used in the past to map SAV with accuracies of about 75% for classes including high-density sea grass, low-density sea grass, and unvegetated bottom (Akins, Wang, and Zhou, 2010; Dierssen *et al.*, 2003; Wolter, Johnston, and Niemi, 2005).

MONITORING WETLAND CHANGES

To identify long-term trends and short-term variations, such as the impact of rising sea levels and hurricanes on wetlands, researchers need to analyze time series of remotely sensed imagery. The acquisition and analysis of time series of multispectral imagery is a difficult task. The imagery must be acquired under similar environmental conditions (e.g., same time of year and sun angle) and in the same or similar spectral bands. There are changes in both time and spectral content. One way to approach this problem is to reduce the spectral information to a single index, reducing the multispectral imagery into one field of the index for each time step. In this way, the problem is simplified to the analysis of time series of a single variable, one for each pixel of the images.

The most common index used is the Normalized Difference Vegetation Index (NDVI), which is expressed as the difference between the red and the near-infrared reflectances divided by their sum. These two spectral bands represent the most detectable spectral characteristic of green plants. This is because the red (and blue) radiation is absorbed by the chlorophyll in the surface layers of the plant (*Palisade parenchyma*) and the near-infrared is reflected from the inner leaf cell structure (*Spongy mesophyll*) as it penetrates several leaf layers in a canopy. Thus, the NDVI can be related to plant biomass or stress, since the near-infrared reflectance depends on the abundance of plant tissue and the red reflectance indicates the surface condition of the plant. It has been shown by researchers that time series of remote sensing data can be

Table 1. High-resolution satellite parameters and spectral bands.*

Sponsor	IKONOS		QuickBird		OrbView-3		WorldView-1		GeoEye-1		WorldView-2	
	Space Imaging	DigitalGlobe	Orbimage	DigitalGlobe	GeoEye	DigitalGlobe						
Launched	Sept. 1999	Oct. 2001	June 2003	Sept. 2007	Sept. 2008	Oct. 2009						
Spatial Resolution (m)												
Panchromatic	1.0	0.61	1.0	0.5	0.41	0.5						
Multispectral	4.0	2.44	4.0	NA	1.65	2						
Spectral Range (nm)												
Panchromatic	525–928	450–900	450–900	400–900	450–800	450–800						
Coastal blue	NA	NA	NA	NA	NA	NA						
Blue	450–520	450–520	450–520	NA	450–510	450–510						
Green	510–600	520–600	520–600	NA	510–580	510–580						
Yellow	NA	NA	NA	NA	NA	NA						
Red	630–690	630–690	625–695	NA	655–690	630–690						
Red edge	NA	NA	NA	NA	NA	NA						
Near-infrared	760–850	760–890	760–900	NA	780–920	770–1040						
Swath width (km)	11.3	16.5	8	17.6	15.2	16.4						
Off nadir pointing (°)	±26	±30	±45	±45	±30	±45						
Revisit time (d)	2.3–3.4	1–3.5	1.5–3	1.7–3.8	2.1–8.3	1.1–2.7						
Orbital altitude (km)	681	450	470	496	681	770						

* From DigitalGlobe (2003), Orbimage (2003), Parkinson (2003), and Space Imaging (2003).

used effectively to identify long-term trends and subtle changes of NDVI by means of principal component analysis (Jensen, 2007; Young and Wang, 2001; Yuan, Elvidge, and Lunetta, 1998).

The preprocessing of multidate sensor imagery, when absolute comparisons among different dates are to be carried out, is more demanding than the single-date case. It requires a sequence of operations, including calibration to radiance or at-satellite reflectance, atmospheric correction, image registration, geometric correction, mosaicking, subsetting, and masking out clouds and irrelevant features. In the preprocessing of multidate images, the most critical steps are the registration of the multidate images and their radiometric rectification. To minimize errors, registration accuracies of a fraction of a pixel must be attained. The second critical requirement for change detection is attaining a common radiometric response for the quantitative analysis for one or more of the image pairs acquired on different dates. This means that variations in solar illumination, atmospheric scattering and absorption, and detector performance must be normalized, *i.e.*, the radiometric properties of each image must be adjusted to those of a reference image (Coppin *et al.*, 2004; Lunetta and Elvidge, 1998).

Detecting changes between two registered and radiometrically corrected images from different dates can be accomplished by employing one of several techniques, including postclassification comparison and spectral image differencing (change detection). In postclassification comparison, two images from different dates are independently classified. The two maps are then compared pixel by pixel. This avoids the difficulties in change detection associated with the analysis of images acquired at different times of the year or day or by different sensors, thereby minimizing the problem of radiometric calibration across dates. One disadvantage is that every error in the individual date classification maps is also present in the final change detection map (Dobson *et al.*, 1995; Jensen, 1996; Lunetta and Elvidge, 1998).

Spectral change detection (spectral image differencing) is the most widely applied change detection algorithm. Spectral change techniques rely on the principle that land cover changes result in changes in the spectral signature of the affected land surface. These techniques involve the transformation of two original images to a new single-band or multiband image in which the areas of spectral change are highlighted. This is accomplished by subtracting one date of raw or transformed (*e.g.*, vegetation indices or albedo) imagery from a second date that has been precisely registered to the image of the first date. Pixel difference values exceeding a selected threshold are considered changed. A change–no change binary mask is overlaid onto the second date image, and only the pixels labeled as having changed are classified in the second date imagery. While the unchanged pixels remain in the same classes as in the first date imagery, the spectrally changed pixels must be further processed by other methods, such as a classifier, to produce a labeled land cover change map. This approach eliminates the need to identify land cover changes in areas where no significant spectral change has occurred between the two dates of imagery (Coppin *et al.*, 2004; Jensen, 1996; Yuan, Elvidge, and Lunetta, 1998). However, to obtain accurate results, radiometric normalization must be applied to one date of imagery to match the radiometric condition of the two dates of data before image subtraction. An evaluation of the spectral image differencing and the post-classification comparison change detection algorithms is provided by Macleod and Congalton (1998).

The spectral change (image differencing) detection methods and the classification-based methods are often combined in a hybrid approach. For instance, spectral change detection can be used to identify areas of significant spectral change, and then postclassification comparison can be applied within areas where spectral change was detected to obtain class-to-class change information. As shown in Figure 3, change analysis results can be further improved by including probability filtering that allows only certain changes and forbids others

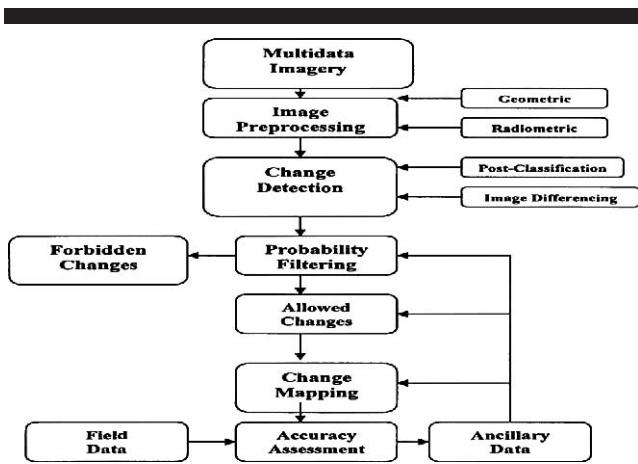


Figure 3. Change detection using probability filters.

(e.g., urban to forest). A detailed, step-by-step procedure for performing change detection was developed by the National Oceanic and Atmospheric Administration's (NOAA's) Coastal Change Analysis Program and is described in Dobson *et al.* (1995) and Klemas *et al.* (1993).

CASE STUDIES

The following two case studies were selected to illustrate and compare the use of practical remote sensing techniques for studying key problems at different wetland sites and to try to answer wetland managers' questions, such as the following: (1) How are urban sprawl and development affecting wetlands in coastal watersheds? (2) How is accelerated local sea level rise changing the vegetation, inundation levels, and hydrology in tidal wetlands? (3) Should one intervene in the hydraulic regime by channel modification to accelerate or delay marsh development in a particular direction?

The case studies do not represent all possible uses of remote sensing in wetlands but are typical of some problems encountered by wetland scientists and managers. The choice of case studies was also based on the author's personal experience.

Remote Sensing Applications Assessment Project (RESAAP)

Managers of NOAA's National Estuarine Research Reserve System (NERRS) have a continuing need to use remote sensing to address typical questions, such as the following: (1) What is the extent of emergent, intertidal, and submerged habitats? (2) How are the emergent, intertidal, and submerged habitats changing? (3) How are suburban sprawl and coastal development affecting reserve watersheds? (4) How have invasive plants affected habitat? (5) How diverse is each NERRS site in terms of habitat types?

While remote sensing was being actively used within NERRS, the multitude of new satellite and aircraft sensors and image analysis techniques that are becoming available make it difficult for research reserve managers to select the

most cost-effective sensing and analysis techniques. Therefore, in 2004, NOAA's NERRS program funded a team of remote sensing experts to compare the cost, accuracy, reliability, and user-friendliness of four remote sensing approaches for mapping land cover, emergent wetlands, and SAV. Four NERRS test sites were selected for the project, including the Ashepoo, Combahee, and South Edisto Basin, South Carolina; Grand Bay, Michigan; St. Jones River and Blackbird Creek, Delaware; and Padilla Bay, Washington (Porter *et al.*, 2006). The research described here was primarily conducted at Delaware's St. Jones River and Blackbird Creek NERRS sites, where wetland changes at the sites and the land cover of their watersheds were studied and mapped.

The Blackbird Creek study site consists of the estuarine and freshwater tidal wetlands within the Blackbird Creek drainage basin and some contiguous wetland areas from two adjacent drainage basins. The study area is approximately 100 km² and covers 19.1 km (11.9 mi.) of the creek. Blackbird Creek is located in southern New Castle County, Delaware, and the upper Blackbird Creek is one component of Delaware's NERRS sites. The upland land use in Blackbird Creek basin is primarily agriculture (51%) and forested (48%), with a small proportion of developed land (1%). Within the wetlands of Blackbird Creek, the amount of direct physical alteration—diking, ditching, channel straightening, and impounding—has been minimal compared to that at many other coastal wetlands in Delaware (Field and Philipp, 2000b).

The Delaware St. Jones River NERRS study site provides a contrast to the Blackbird Creek site in several ways. The total area of the study site is approximately 80 km², and it covers 14.3 km (8.9 mi.) of the main river channel. The amount of agriculture in the St. Jones River basin is 51%, forested area is 38%, and developed land equals 11%, including higher-density residential areas and commercial or industrial developments. The St. Jones River has been subjected to much direct human manipulation. The natural course of the river's main channel has been straightened, and parallel grid ditches were dug in a portion of the wetlands for mosquito control. These hydrologic alterations have undoubtedly affected the wetland ecosystem structure and functions in this river.

The RESAAP team also included scientists from the University of South Carolina and NOAA who were performing similar studies of emergent wetlands and SAV at three other NERRS sites (Porter *et al.*, 2006). Results were compared to determine which imagery and analysis approach should be recommended for use at other NERRS sites.

The four remote sensing systems evaluated were the hyperspectral airborne imaging spectrometer for applications (AISA), an aerial multispectral (ADS 40) digital modular camera, the IKONOS (or QuickBird) high-resolution satellite, and Landsat TM. A comparison of approximate data acquisition costs is shown in Table 2. The high-resolution imagery per square kilometer of coverage is much more expensive than the medium-resolution imagery.

Completed in 2006, this study found that aerial hyperspectral image analysis is too complicated for typical NERRS site personnel and the imagery is too expensive for large NERRS sites or entire watersheds. Furthermore, it was difficult to discriminate wetlands species even with hyperspectral imag-

Table 2. *Imagery acquisition costs (Porter et al., 2006).*

Description	Resolution (m)	Other Features	Cost (\$/km ²)
Digital camera imagery, ADS40	0.3	Cell area = 2.3 × 2.3 km	330
Aerial hyperspectral, AISA	2.3	Swath width = 600 m; spectral channels = 35 (0.44–0.87 μm)	175
High-resolution satellite, IKONOS	1–4	Swath width = 13 km	30
Medium-resolution satellite, Landsat TM	30	Swath width = 180 km	0.02 (\$600 per scene)

ery (Porter *et al.*, 2006). Due to different sun angles for each flight strip, a separate atmospheric correction had to be implemented for each strip. Also, the aircraft roll due to wind conditions produced uneven swaths.

In the NERRS study, the highest accuracy for mapping clusters of different plant species over small critical areas was obtained by visually analyzing orthophotos produced by airborne digital cameras. The visual interpretation was performed after image segmentation and with the help of field training sites visited before and after the interpretation process. For larger sites, combining IKONOS and Landsat TM proved cost-effective and user-friendly. The Landsat TM imagery was used to map land cover for the large site or entire watershed, and the IKONOS high-resolution imagery was used for detailed mapping of critical NERRS areas or those identified by Landsat TM as having changed. A particularly effective technique developed by the team is based on using biomass change as a wetland change indicator (Porter *et al.*, 2006; Weatherbee, 2000).

Monitoring Accelerated Local Sea Level Rise in Wetlands

The primary objectives of this project were to study changes at a unique Delaware Bay tidal wetland site, which faces an accelerated sea level rise due to a canal breach, and to show how remote sensors and related techniques can be used for studying the impact of sea level rise and man-made influences on coastal wetlands. The improved understanding of the processes occurring at this rapidly changing site will help wetland managers decide whether to intervene in the hydraulic regime by channel modification to accelerate or delay marsh development in a particular direction.

The study site was the Milford Neck Conservation Area (MNCA), which is located along the southwestern shore of Delaware Bay. It contains 10,000 acres of tidal marsh and 9 mi. of shoreline. The complex, dynamic landscape of this site is characterized by a transgressing shoreline, extensive tidal wetlands, island hammocks, and upland forests. A canal (Greco's Canal) separates the site from a narrow barrier beach along Delaware Bay. Recent changes in the shoreline and tidal marsh have resulted in dramatic habitat conversion and loss that may have significant immediate and long-term impacts on the biologic resources and ecologic integrity of the MNCA (Field and Philipp, 2000b).

The barrier beach of the MNCA was breached during the winter of 1985–86, making a direct connection between Delaware Bay and Greco's Canal. Before the breach, the hydraulic regime of the marsh west of Big Stone Beach was controlled through the canal to the Mispillion River far to the

south (Figure 4). The breach through the barrier beach resulted in a shorter and direct linkage of the marsh to the tidal forcing of Delaware Bay. This has changed the tide regimes experienced in the various sections of the marsh and the resulting patterns of tide marsh vegetation.

A newly established gravel sill in the mouth of the canal at the breach seems to regulate the interior hydrology by establishing base water levels in Greco's Canal, which are higher than low water levels in the bay. Continuing beach overwash and continuing westward migration of the beach provide a source of sand and gravel to maintain and enlarge the sill. The sill is growing northward in the canal in response to the large hydraulic head established during spring tide and storms in the bay. During ebb tide, the sill can be only slowly eroded because of the relatively small hydraulic head above the sill to drive drainage from a lagoon (Field and Philipp, 2000a). As shown in Figure 4, AISA hyperspectral and IKONOS satellite imagery was used to determine that in just 2 years, from 1999 to 2001, the area of open water plus scoured mud bank increased by about 50% due to the increased tidal flushing after the canal breach. Since the canal breach allowed tidal waters to flow directly into the marshes, the average width of some major creeks changed from 5.1 to 7.3 m and the bank widths affected by tidal scouring increased from about 9.1 to 16.2 m. On the right sides of the images, you can clearly see Greco's Canal and the breach connecting it to Delaware Bay (Field and Philipp, 2000a).

At the MNCA site, there has been a general trend for high salt marsh to be replaced by lower salt marsh vegetation, mudflats, and open water. Thus, there are decreases in the extent of salt hay cover (*S. patens* and *Distichlis spicata*) and increases in the expanse of open water, mudflats, *Spartina alterniflora*, and *Phragmites australis*. The less desirable common reed (*P. australis*) has been expanding despite treatments with herbicides since 1999. Large areas of tidal marsh NW of the breach have become permanently inundated and converted into subtidal marsh. Analysis of Landsat TM images for 1984 and 1993 shows that the area of open water west of Greco's Canal has increased from about 40 to 160 ha, with a corresponding loss of highly productive *S. alterniflora* marsh. The area of open water and mudflats lying to the east of the canal has also increased dramatically during this period. Vegetation bordering natural ponds within the marsh and near the interface of marsh and upland forest shifted toward a less diverse, more salt-tolerant community (Field and Philipp, 2000a).

The general direction of the vegetation changes was not surprising; *i.e.*, uplands changed to high marsh, high marsh changed to low marsh, and low marsh was in many places inundated to produce open water and mudflats. What was

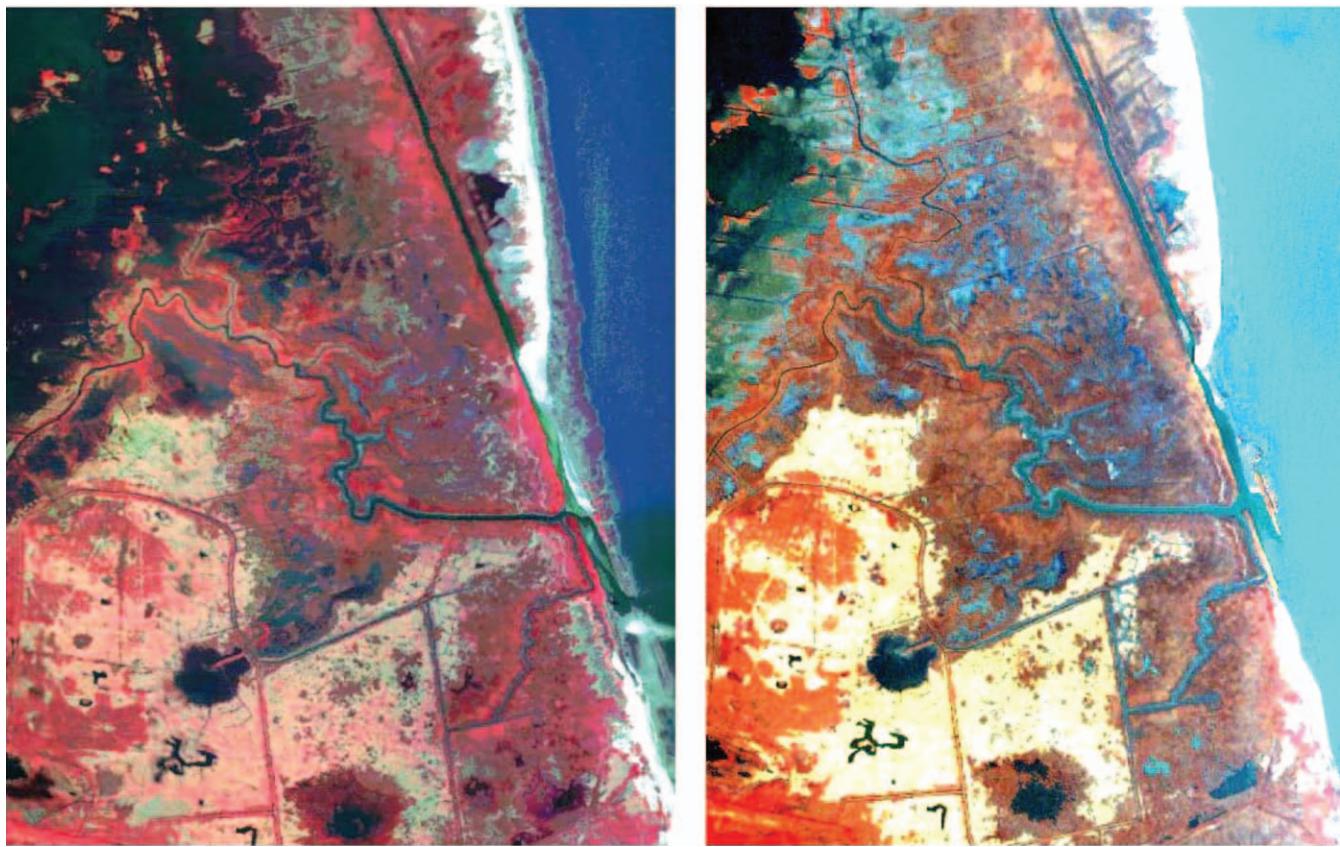


Figure 4. Images showing vegetation, inundation, and hydrologic changes at the MNCA site between 1999 and 2001. (Left) An AISA hyperspectral image of 1-m resolution obtained on 18 September 1999. (Right) An IKONOS satellite image of merged 1–4-m resolution captured on 24 August 2001. (Modified from Field and Philipp, 2000a.)

surprising was the rapid pace at which these changes took place as the “accelerated” local sea level kept rising.

SUMMARY AND CONCLUSIONS

The advent of new satellite and airborne remote sensing systems having high spectral (hyperspectral) and spatial resolutions, has improved our capacity for mapping upstream wetlands, salt marshes, and general coastal vegetation. Using hyperspectral imagery and narrow-band vegetation indices, researchers have been able to identify some wetland species and to make progress on estimating biochemical parameters of wetland vegetation, such as water content, biomass, and leaf area index. The higher spatial resolution makes it possible to study small critical sites, including rapidly changing or patchy upstream wetlands.

The integration of hyperspectral imagery and LIDAR-derived data has improved the accuracy of mapping salt marsh vegetation and can also provide information on marsh topography, beach profiles, and bathymetry. High-resolution synthetic aperture radar allows researchers to distinguish between forested wetlands and upland forests.

Since wetlands and estuaries have high spatial complexity and temporal variability, satellite observations must usually be

supplemented by aircraft and field data to obtain the required spatial, spectral, and temporal resolutions. Similarly, mapping coral reefs and SAV requires high-resolution satellite or aircraft imagery and, in some cases, hyperspectral data.

To identify long-term trends and short-term variations, such as the impact of rising sea levels and hurricanes on wetlands, researchers need to analyze time series of remotely sensed imagery. The images must be acquired under similar environmental conditions (e.g., same time of year and sun angle) and in similar spectral bands. In the preprocessing of multidate images, the most critical steps are the registration of the multidate images and their radiometric rectification. To minimize errors, registration accuracies of a fraction of a pixel must be attained. To detect changes between two corrected images from different dates several techniques can be employed, including postclassification comparison and spectral image differencing.

The two case studies presented in this paper clearly illustrate the practical aspects of wetland remote sensing. In the NERRS study, the highest accuracy for mapping clusters of different plant species over small critical areas was obtained by visually analyzing orthophotos produced by airborne digital cameras. To achieve cost-effectiveness, Landsat TM imagery was used to map land cover for large sites or entire watersheds,

and IKONOS high-resolution imagery was used only for detailed mapping of critical NERRS areas or those identified by Landsat TM as having changed. The changes observed in the satellite imagery include land cover change, buffer degradation, wetland loss, biomass change, wetland fragmentation, and invasive species expansion.

In a study of changes at a unique Delaware Bay tidal wetland site, which faces an accelerated sea level rise due to a canal breach, satellite and airborne digital sensors of 1- and 2-m ground resolution enabled researchers to track annual changes in the details of the vegetation patterns and hydrologic networks. For instance, by comparing AISA hyperspectral imagery with 1- and 4-m resolution IKONOS images acquired in October 2000 and September 2001, respectively, it was possible to measure major changes in the width of tide channels, width of scoured creek banks, areas of open water, and length of open water (Figure 4). Analysis of Landsat TM images, acquired over a decade, were used to determine that the area of open water to the west of Greco's Canal had increased from 40 to 160 ha. (Field and Philipp, 2000a). The case studies showed that satellite and aircraft remote sensors, supported by a reasonable number of site visits, are suitable and practical for mapping and studying coastal wetlands, including long-term trends and short-term changes of vegetation and hydrology. Some practical recommendations can be made, based on the results of the case studies:

- (1) The cost *per* square kilometer of imagery and its analysis rises rapidly with the shift from medium- to high-resolution imagery. Therefore, large wetland areas or entire watersheds should be mapped using medium-resolution sensors (e.g., Landsat TM at 30 m), and only small, critical areas should be examined with high-resolution sensors (e.g., IKONOS at 1–4 m).
- (2) Multispectral imagery should be used for most applications, with hyperspectral imagery reserved for difficult species identification cases, larger budgets, and highly experienced image analysts.
- (3) Airborne digital camera imagery is not only useful for mapping coastal land cover but also helpful in interpreting satellite images.
- (4) The combined use of LIDAR and hyperspectral imagery can improve the accuracy of wetland species discrimination and provide a better understanding of the topography, bathymetry, and hydrologic conditions.
- (5) High-resolution imagery is more sensitive to within-class spectral variance, making separation of spectrally mixed land cover types more difficult. Therefore, pixel-based techniques are sometimes replaced by object-based methods, which incorporate spatial neighborhood properties (Shan and Hussain, 2010; Wang, Sousa, and Gong, 2004).

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