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Assessment of multi-resolution image data for mangrove leaf area index mapping

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ABSTRACT

The increasing development pressures from urbanization, aquaculture and tourism on worldwide coastal environments and the ecosystem services that mangroves provide make it essential to monitor and manage these environments more effectively. Measuring and monitoring mangrove structure through variables like Leaf Area Index (LAI) is an essential part of this action. This study investigated the effects of different mangrove environmental settings, satellite image spatial resolutions, spectral vegetation indices (SVIs) and mapping approaches for LAI estimation. We compared and contrasted the ability of WorldView-2 (WV-2), ALOS AVNIR-2 (AVNIR-2) and Landsat TM (TM) image data (2 m, 10 m and 30 m pixel sizes, respectively), to estimate LAI through regression analysis at sites in Moreton Bay (Australia) and Karimunjawa Island (Indonesia). We also investigated the effect of different pixel averaging windows (3 × 3, 5 × 5, and 7 × 7 pixels) and multi-resolution segmentation scale parameters (10, 20, 30, 40 and 50) applied to the WV-2 image for LAI estimation. The results showed that LAI estimation using remote sensing data varies across sites and sensors. Estimation of LAI in this study was influenced by the local spatial variation of mangrove phenological stages and canopy cover. The regression analyses showed significant coefficient of determination (R^2) values ranging from 0.50 to 0.83 across different sensors (TM, AVNIR-2, WV-2), segmentation scales (10, 20, 30, 40, 50) and SVIs (SR, NDVI, SAVI, EVI). The sensor and SVIs assessment identified the ALOS AVNIR-2 and NDVI as the optimal estimators of LAI, with $R^2 = 0.83$, RMSE = 0.54 for Moreton Bay, and $R^2 = 0.82$, RMSE = 1.31 for Karimunjawa Island. The optimum image pixel size for estimating LAI was related to the average canopy size (about 10 m in diameter) and the field sampling size (10 m). Image segmentation significantly increased the LAI estimation accuracy by approximately 14% for both sites. The findings of this study provide an understanding of the relationship between image spatial resolution, field sampling size and spatial variation of mangrove vegetation for estimating LAI. These findings can be potentially used as a guide for selecting the optimum imagery for LAI estimation.

1. Introduction

Leaf area index (LAI) is one of the most important biophysical parameters for assessing mangrove forest health (Giri, Pengra, Zhu, Singh, & Tieszen, 2007; Heumann, 2011; Jensen et al., 1991). It is defined as the one-sided leaf area per unit surface area (m^2/m^2), and therefore is a dimensionless number (Addink, deJong, & Pebesma, 2007; Green, Mumby, Edwards, Clark, & Ellis, 1997; Lymburner, Beggs, & Jacobson, 2000; Pierce & Running, 1988). The importance of LAI in vegetation studies is well-recognised. It is an indicator of ecological processes (rates of photosynthesis, transpiration and evapotranspiration) (Pierce & Running, 1988), net primary production (Clough, Ong, & Gong, 1997; Meyers & Paw, 1986, 1987) and rates of energy exchange between plants and the atmosphere (Gholz et al., 1991). LAI can be used to predict future growth and yield (Gholz, 1982) and assists in monitoring changes in canopy structure due to pollution and climate change (Fassnacht, Gower, MacKenzie, Nordheim, & Lillesand, 1997; Gholz et al., 1991). Due to its significance in describing a fundamental property of the plant canopy in its interaction with the atmosphere and solar radiation (Bréda, 2008), the ability to estimate LAI provides a valuable means to understand and estimate the physical condition of mangroves (Kovacs, King, Flores de Santiago, & Flores-Verdugo, 2009). Collecting and assessing change in mangroves is essential as at least 35% of the global mangrove area was reported lost during the past two decades (FAO, 2007), exceeding losses reported for tropical rain forests and coral reefs (Valiela, Bowen, & York, 2001). Predictions suggest that in the next 100 years, about 30–40% of coastal wetlands will be lost (McFadden, Nicholls, & Penning-Rowsell, 2007), including 100% of mangrove forest (Duke et al., 2007) if the present rate of loss continues.

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The alarming status of global mangrove forest losses corroborates the need to develop cost-effective and accurate techniques for rapid mangrove LAI mapping. Direct measurements of LAI in mangroves yield very accurate results. However, it is difficult to access mangrove forest due to the mangrove root systems and tidal fluctuation, it is labour intensive and costly in terms of time and money, and some of the methods are destructive (Bréda, 2008; Green et al., 1997). As an alternative, indirect and spatially explicit LAI estimation from remote sensing data provides a practical method to repeatedly map LAI from local to global scales (Fang & Liang, 2008). Studies have indicated the successful implementation of optical remote sensing data for mangrove LAI mapping from various sensors; Landsat TM or ETM+ (Díaz & Blackburn, 2003; Green et al., 1997; Ishil & Tateda, 2004; Ramsey & Jensen, 1996), SPOT XS (Green et al., 1997; Ramsey & Jensen, 1996), AVHRR (Ramsey & Jensen, 1996), ASTER (Jean-Baptiste & Jensen, 2006), IKONOS (Kovacs, de Santiago, Bastien, & Lafrance, 2010; Kovacs, Flores-Verdugo, Wang, & Aspden, 2004; Kovacs, Wang, & Flores-Verdugo, 2005), QuickBird (Kovacs et al., 2010; Kovacs et al., 2009), CASI (Green, Mumby, Edwards, Clark, & Ellis, 1998), Leica-ADS40 (Kovacs et al., 2010) and ALOS PALSAR (Kovacs et al., 2013).

The estimation of mangrove LAI using optical remote sensing data has been based on empirical or semi-empirical statistical relationships formulated between in-situ LAI measurements and the image pixel values, from at-surface spectral reflectance or in the form of spectral vegetation indices (SVIs) (Jean-Baptiste & Jensen, 2006; Kovacs et al., 2009; Laongmanee, Vaiphasa, & Laongmanee, 2013). The indices are designed to enhance the sensitivity of the spectral reflectance contribution of vegetation while minimizing the soil background reflectance or atmospheric effects (Fang & Liang, 2008; Huete, 2012). These empirical statistical relationships are then used to estimate the distribution of LAI in an image. For example, the linear regression applied to the normalised difference vegetation index (NDVI) derived from SPOT XS had a high correlation with LAI ($R^2 = 0.74, p < 0.001, n = 29$) in South Caicos and Caicos Bank, British West Indies (Green et al., 1997). Significant relationships were also found between LAI and the simple ratio (SR) and the NDVI based on QuickBird imagery and a linear regression ($R^2 = 0.63$ and 0.68, respectively, $p < 0.0001$, $n = 225$) in the Teacapán–Agua Brava–Las Haciendas estuarine mangrove system, Mexican Pacific (Kovacs et al., 2009).

The recent development of optical remote sensing technology allow exploration of a wide range of optical image datasets, with pixel sizes ranging from sub-metre to hundreds of metres, and spectral information ranging from narrow hyper-spectral bands to broadband multispectral images. At the same time, the rapid development of image processing techniques suitable for high spatial resolution image data, e.g. geographic object-based image analysis (GEOBIA), has shifted the way image-based mapping is performed (Blaschke, 2010; Blaschke & Strobl, 2001). As opposed to the conventional pixel-based methods, GEOBIA produces meaningful objects that are represented by a cluster of neighbouring homogenous pixels through image segmentation based on the spectral information and local pattern or textural information (Baatz & Schape, 2000; Benz, Hofmann, Willhauck, Lingenfelder, & Heynen, 2004; Blaschke & Strobl, 2001). One of the advantages of image segmentation in GEOBIA is its flexibility to adjust the scale of the targeted objects (Benz et al., 2004; Trimble, 2011). Currently, there are a very limited number of GEOBIA studies investigating the effects of image segmentation scales on LAI mapping.

Based on the premise that the selection of an appropriate image spatial resolution is essential for the successful application of remote sensing (Phinn, Menges, Hill, & Stanford, 2000; Woodcock & Strahler, 1987), this study assessed the effects of different image spatial resolutions and pixel aggregation (i.e. image segmentation) on estimating LAI in two different mangrove habitats using spectral vegetation indices. The main objectives of this study were to investigate: (1) whether different remote sensing data (TM, AVNIR-2, WV-2) affect the estimation of LAI in different mangrove habitats (i.e. tropic and sub-tropic); (2) which of the remote sensing data sets and SVIs (SR, NDVI, SAVI, EVI) provide the most accurate estimation of LAI, and (3) whether GEOBIA improves LAI estimation compared to pixel-based models.

2. Study area

The research was conducted in two mangrove ecosystems; the mouth of Brisbane River, northern Moreton Bay, South East Queensland, Australia and Karimunjawa National Park, Central Java, Indonesia (Fig. 1). The mangrove extent in Fig. 1 was mapped using object-based image analysis through hierarchical rule sets applied to WV-2 image; detail explanation of the methods is provided in Kamal, Phinn, and Johansen (2015). The first site (between 153°3′41″–153°11′20″ E and 27°19′41″–27°25′31″ S) is a sub-tropical lowland area, which includes Whyte Island, Fisherman Island and Boondall wetlands, approximately 15 km northeast of Brisbane city. It is one of Australia's premier wetlands and a Ramsar Convention listed wetland, with extensive stands of mangroves (Environment Australia, 2001). Mangroves in Moreton Bay are dominated by *Avicennia marina*, which comprise ~75% of the entire mangrove community (Dowling & Stephen, 2001). Some *Rhizophora stylosa* individuals are found sporadically as a mid-storey between *Avicennia* stands. Several patches of uniform *Ceriops tagal* stands are found near the creeks of Fisherman Island and Boondall wetlands, and *Aegiceras corniculatum* are found mostly as understorey (Duke, 2006). Distinct structural zonations are noticeable in this area from the saltmarsh area, through the mangroves to the water. The progression is open scrub formation (S3), followed by low-closed forest (I4), and finally closed forest (M4), according to the forest structure classification by Specht, Specht, Whelan, and Hegarty (1995).

The second site (between 110°24′10″–110°30′10″ E and 4°47′48″–5°50′12″ S) is located in the Java Sea between Java and Kalimantan Islands (Fig. 1). Mangroves in Karimunjawa National Park exist mainly in the fringing area on the western side of the two main islands; Karimunjawa and Kemujan. According to a Karimunjawa National Park Office report (BTNK, 2011), there are 45 mangrove species in this area (27 true mangroves and 18 mangrove associates), *R. stylosa* being the most dominant mangrove species. Although it is less apparent when compared to Moreton Bay mangroves, three different mangrove structural formations were recognised from the land to the seaward margin. The first landward formation is dominated by low multi-stem stands (VL4) of *C. tagal* and *Lumnitsera racemosa*. The middle formation is the single and multi-stem low-closed forest (I4) of highly mixed formations of *C. tagal*, *Lumnitsera* sp., *Rhyzophora* sp. and *Bruguiera gymnorrhiza*. Lastly, closer to the shoreline, is a formation of multi-stem closed forest (M4) of *Rhizophora mucronata* and some individual *Bruguiera gymnorhiza* and *Xylocarpus granatum*.

In the context of this study, the Moreton Bay site is dominated by homogenous mangrove species stands, while the Karimunjawa site represents heterogeneous species stands. At both sites, mangrove zonations were noticeable at different distances from the coastline towards the landward limit of the mangroves. These locations were selected to understand the variation in LAI at different mangrove vegetation structure and environmental setting and to investigate the optimum pixel size, in relation to the field sampling scheme, for estimating LAI at multiple sites.

3. Data and methods

3.1. Fieldwork and LAI measurements

Fieldwork was conducted in April 2012 at the Moreton Bay sites and in July 2012 on Karimunjawa Island. The selection of these dates was aimed to resemble the season in which the WV-2 images were acquired (i.e. autumn [April 2011] and dry season [May 2012], respectively). Twenty-three field transects perpendicular to the shoreline were laid out at both sites (Fig. 1) to collect structural measurements of