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Quantifying isolation effect of urban growth on key ecological areas

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ABSTRACT

Urbanization may threaten multiple ecosystem processes as well as biodiversity in ecological reserves due to the loss, fragmentation and degradation of habitats in unprotected areas, making it necessary to measure the interference effect of urban growth on the protected areas. This study develops an urban isolation index (UII) to quantify the isolation effects of land-use conversion on key ecological areas (KEAs) and accounts for urban patch size, morphology, location, and quality loss. Based on the five remote sensing Landsat images taken between 1980 and 2005, UII is applied to analyze the spatio-temporal dynamics of the isolation effects caused by rapid urbanization in Shenzhen, China. The results showed that the habitat isolation constantly intensified during the urbanization process. We identified spatially explicit information of critical urban domains that exerted an interference effect on KEAs. There are considerable differences of isolation effects resulting from different urban growth types. Urban edge expansion caused the greatest habitat isolation in Shenzhen. The outlying and infilling urban patches contributed minor isolation effects to the KEAs. This paper furthers the understanding of the ecological effects of urbanization processes and provides a spatially explicit identification for ecological conservation and restoration in rapidly urbanized regions.

1. Introduction

The urban population of the world grew almost exponentially during the late 18th century. Today, half of the world's population resides in urban areas. It is predicted that the global urban population will increase to 69% by 2050 (UNPFA, 2008). Urbanization, a complex process that transforms landscapes formed by rural life styles into urban landscapes (Antrop, 2000), is becoming increasingly universal and irresistible throughout the world. Urbanization dramatically shapes new landscape patterns and significantly influences local and global ecosystem functions and dynamics. In most urban areas, urban spatial diffusion can fragment, isolate, and degrade natural habitats (Alberti, 2005). These negative effects on habitats not only involve the direct extinction and reduction of native wildlife but also alter multiple ecological processes, such as animal movement, seed dispersal, genetic flow and nutrient cycling (Harrison and Bruna, 1999; Alberti, 2005; Crooks and Sanjayan, 2006; Grimm et al., 2008), and, ultimately, influence ecosystem services and human well-being (Alberti, 2005). Furthermore, urban development is facing the dilemma of urban land intransigence or natural habitat degradation. The only way to face this challenge is to develop sustainable practices, i.e., economically, socially and ecologically acceptable solutions, to mitigate the conflicts between urban development and ecological conservation in urbanized areas (Barot et al., 2012).

Ecological engineering is defined as the design of sustainable ecosystems that integrate human society with the natural environment for the benefit of both (Mitsch, 2012). Its goals involve the creation or restoration of ecosystems that can provide sustainable services for humans and other forms of life (Mitsch, 2012). Since the 20th century, establishing protected areas has become the fundamental strategy of regional ecological conservation and restoration (Howard et al., 2000). Ecological reserves may be the most direct and effective measure to maintain species persistence and biodiversity, especially in urban areas. During

Abbreviations: UII, urban isolation index; CUII, cumulative urban isolation index; NPP, net primary productivity; PSI, patch shape index; KEA, key ecological areas; SEZ, special economic zone; ERA, ecosystem restoration assessment.

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urbanization processes, however, ecological reserves are likely to cause ecological isolation among protected areas if the surrounding unprotected vegetation habitats undergo anthropogenic disturbances, and can even lead to a conversion to human-dominated landscapes (Grumbine, 1994). In other words, from the viewpoint of ecosystem integrity, whether reserves can provide sustained and effective protection for organisms depends not only on the protected areas themselves but also on the natural context outside of the protected boundaries (Hansen and DeFries, 2007). This is largely because some of the key areas excluded from the protected domains are important to protect wildlife from exposure to a hostile landscape matrix and to maintain nutrient flows, organism movements and population processes (Grumbine, 1990). It is therefore essential and implicative for regional-scale ecological conservation and restoration, especially in a large-range urbanization context, to evaluate the isolation effect of urban growth on the ecological reserves in a quantitative and comprehensive manner.

The quantification method of landscape fragmentation and connectivity has always been a hot topic for ecologists (Lindenmayer et al., 2008). Early proposed landscape metrics, such as the patch density, mean patch size, perimeter-area ratio and fractal index, can only capture the separate aspects of fragmentation (Davidson, 1998). Recently, some new metrics, such as the landscape division, splitting index and effective mesh size, have been developed to characterize landscape fragmentation from a geometric perspective (Jaeger, 2000; Moser et al., 2007). Landscape connectivity is defined as the degree to which the landscape facilitates or impedes the dispersal movement of species among habitat patches that exist in the landscape (Taylor et al., 1993). Graph theory is thought to be a promising methodology in the assessment of landscape connectivity (Urban et al., 2009). Some scholars have proposed a series of graph-based metrics to quantify the network connectivity related to species dispersal processes (e.g., Urban and Keitt, 2001; Pascual-Hortal and Saura, 2006; Saura and Pascual-Hortal, 2007; Yu et al., 2013). In general, the metrics used to assess landscape fragmentation and connectivity are well developed; some of them have even been successfully applied to biological conservation and ecological restoration (e.g., Cabeza, 2003; Zetterberg et al., 2010; Yu et al., 2012). However, most of the metrics concentrate on the fragmentation features of a single landscape type and thus cannot reflect the interaction among landscape types during land-use conversion processes.

Habitat fragmentation may cause functional isolation due to the weak exchange of genes for both individuals and populations, which explains why fragmented habitats often contain fewer species than contiguous habitats (Forman et al., 1976). Habitat isolation related to interference effects is a more complex process than fragmentation in the context of urbanization. Recently, two metrics, the insulation degree (ID) (Su et al., 2010) and urbanization isolation effect (UIE) (Ng et al., 2011), were proposed to examine the isolation effect caused by urban areas. These metrics were described by the distance of urban patches to the natural patch and the area of urban patches within a specified radius. In the current study, we developed a new urban isolation index (UII) by comprehensively considering the interaction effects of urban size, morphology and quality losses on key ecological areas (KEAs) during land-use conversion processes. We evaluated the spatio-temporal dynamics of the isolation effect of urban growth on KEAs in Shenzhen, an area of China that has been rapidly urbanized. Our objective is to quantitatively answer two questions: (1) What degree of isolation do KEAs undergo during the urbanization process? (2) Which type of urban growth causes the most serious habitat isolation?

2. Material and methods

2.1. Study area

Shenzhen is a coastal city in southern China, neighboring Dongguan and Huizhou in the north and Hong Kong in the south, and is flanked by the Daya Gulf in the east and the Pearl River Estuary in the west. The city consists of a special economic zone (SEZ), Bao'an District and Longgang District (Fig. 1). It lies between 22°26′ N and 22°51′ N latitude and between 113°45′ E and 114°37′ E longitude, and its total area is approximately 2020 km^2. With a subtropical marine climate, the annual average temperature, precipitation and sunshine hours are 22.4 °C, 1933.3 mm and 2020 h, respectively. The rainy season is mainly concentrated from April to September. The dominant vegetation is an evergreen broad-leaved mixed forest. There are 1889 types of wild vascular plants, among which 22 belong to the nationally rare and endangered plants list (Xing and Yu, 2000). There are five types of nationally protected Class I wild animals and 43 nationally protected Class II wild animals (SWCA, 2009).

Shenzhen was municipalized in 1979 and established as the first SEZ in China in 1980. Over the past 30 years, it has evolved from a border fishing village into a metropolitan region with a population of 10.35 million and a GDP of 951.10 billion Yuan in 2010 (SSB, 2011). Meanwhile, Shenzhen has experienced significant land-use changes. In 1980, urban lands and forests occupied 0.63% and 38.67% of the whole city area, respectively. By 2005, urban land had increased to 33.52% and forests had decreased to 29.81% of the total area (see the remote sensing Landsat data in Section 2.2). Because of the habitat fragmentation caused by this urban growth, biodiversity has only remained in isolated mountain areas. In 2005, the Shenzhen government promulgated a legally binding regulation stating that KEAs need to provide a survival environment for wild species, maintain ecosystem processes and regional biodiversity, and promote environmental quality for human well-being. KEAs contain (Fig. 1): 1) water source protection areas, scenic spots, nature reserves, and primary farmland protection areas, forests and country parks; 2) mountains and woodlands with a slope over 25° and uplands with an elevation over 50 m; 3) main trunk rivers, reservoirs and wetlands; 4) green space for the maintenance of ecosystem integrity; and 5) peninsulas and other ecologically important coastal areas. However, the impacts of urban growth on these ecological areas have not undergone scientific assessment until now.

2.2. Data processing

We used five cloud-free Landsat images (including a multi-spectral scanner (MSS) satellite image with a resolution of 79 m × 79 m from 1980; three thematic mapper (TM) satellite images with a resolution of 30 m × 30 m from 1988, 1994 and 2005; and an enhanced thematic mapper plus (ETM+) satellite image with a resolution of 30 m × 30 m from 2000) to produce the land-use thematic maps. Other auxiliary data included topographic maps at a 1:50,000 scale from the 1960s and 1970s, an administrative boundary map at a 1:100,000 scale from 2005, and aerial photographs with high resolution and field survey data to carry out the geometric correction and land-use classification.

We obtained 389 evenly distributed ground control points (GCPs) with exact locations and land cover information to perform the geometrical correction and classification. Different land-use types were categorized using both unsupervised classification and supervised classification algorithms. Based on the normalized