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Assessing the conservation effects of nature reserve networks under climate variability over the northeastern Tibetan plateau

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ABSTRACT

To evaluate the effectiveness of China's largest nature reserve network in providing and protecting habitats, we performed a series of studies consisting of field investigation, model simulation, remote sensing and GIS analysis by using a before-after-control-impact (BACI) design. Data from the non-protected areas (non-PAs) surrounding the nature reserve network served as a baseline for comparison. We found that some indexes of ecological restoration in the nature reserve network were superior to those in the non-PAs, such as average annual vegetation coverage and net primary production. After the establishment of the nature reserve network, ecosystem water retention service increased by 72% compared to those in the non-PAs. In the non-PAs, intensified or new degradation occurred due to overgrazing and over-exploitation. In recent years, the wetting and warming climate has been the main driver of ecological restoration in the region. The establishment of the nature reserve networks and the implementation of conservation projects both promote the improvement of local ecosystems. Therefore, for alpine regions under climate variability, we should protect entire regions rather than specific areas, and management efforts should focus on long-term sustainable conservation rather than emergency rescue.

1. Introduction

Protected areas (PAs) are crucial for the long-term conservation of natural areas and their associated ecosystem services and cultural significance because they could provide safe haven for species threatened by human activities that lead to habitat degradation or loss (Gaston et al., 2008; Joppa et al., 2008; Radeloff et al., 2010). PAs represent important core 'units' for in situ conservation (Pettorelli et al., 2012). Although PAs have proven to be effective in protecting species from human threats, many species might shift their distributions to be outside existing protected areas following climate change (Araújo et al., 2011; Alagador et al., 2014). Current PAs are expected to remain important for future conservation efforts under a changing climate (Hole et al., 2009; Johnston et al., 2013). Therefore, PAs not only provide sustainable management tools that help protect valued features from the processes that threaten them but also help mitigate the impacts of climate change (Araújo et al., 2011).

Governments and non-governmental organizations have invested billions of dollars to conserve biodiversity and habitats, especially those located in biodiversity hotspots of species structure and richness (Myers et al., 2000; Brooks et al., 2006). Despite increases in conservation efforts, the loss of biodiversity continues (Rands et al., 2010). Although habitat degradation, fragmentation, and destruction have all driven recent biodiversity loss, climate change is projected to be a major driver of extinction throughout the 21st century (Thomas et al., 2004; Pereira et al., 2010; Bellard et al., 2012). Due to the widespread loss and fragmentation of habitats, some areas may become climatologically unsuitable for many species (Walther, 2010). Therefore, a better understanding of the interactions between climate change and land use change, as well as their effects on biodiversity conservation strategies inside these PAs, is needed (Hannah et al., 2007; Mazaris et al., 2013; Regos et al., 2016). This issue has been brought into sharper focus by the escalating threats to PAs as a result of climate change and by the debate about whether PAs remain relevant in periods of rapid biophysical and social change (Dunlop and Brown, 2008; Hannah et al., 2007; Shadie and Epps, 2008).

Under the projected climate change scenarios, studies on the effectiveness of PAs are becoming increasingly important, especially because areas covered by the PAs increase rapidly (Soutullo, 2010). Recent studies have focused on individual indicator or comprehensive indices to assess the rationality of PAs, the effectiveness of management strategies, the factors affecting PAs, and the economic and social

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impacts of PAs (Hole et al., 2009; Fan et al., 2012; Zheng et al., 2012; Zhang et al., 2016). Thus, it is important to evaluate whether the PAs reach their long-term conservation objectives (Regos et al., 2016) and whether they will remain effective in providing habitat to protect biodiversity (Hole et al., 2009). Comparing changes of ecological indicators inside and outside PAs has proved to be a practical method of assessing the protection effectiveness (Johnston et al., 2013; Coetzee et al., 2014; Kallimanis et al., 2015). Globally, species richness is 10.6% higher and abundance is 14.5% higher in samples from inside PAs compared to samples from outside PAs (Gray et al., 2016). In addition, landscape metrics like the number and density of patches, and the largest patch index have been used to quantify ecosystem structural and functional of PAs, serving as indicators of conservation (Drakou et al., 2011; Sowinska-Swierkosz and Soszynski, 2014).

As one of the most important areas with PAs, China has established 2740 nature reserves that cover approximately 14.8% of the nation's area, of which 428 are national nature reserves till 2016, for the long-term conservation and their associated ecosystem services and cultural significance. Regular evaluation of the relevance and efficiency of management actions in PAs is needed (Pettorelli et al., 2012). Thus, it is important to monitor the dynamics of ecosystem structure and functions in the PAs and assess the contributions of the PAs in providing habitat to protect biodiversity under a changing climate. In this study, we endeavor to respond to the following questions: 1) How to accurately monitor the dynamics of ecosystem structure and functions to assess conservation effects of nature reserve networks on ecosystems? And 2) Do nature reserve networks consistently provide high-quality habitats in an era of rapid climate variability and increased human activity?

2. Study area

The Sanjiangyuan region with a total area of 0.36 million km^2 is the source region for the Yangtze River (YR), the Yellow River (HR), and the Lancang-Mekong River (LM). It is located in the northeastern part of the Tibetan Plateau and is one of the most important biodiversity hotspots in China. About 0.66 million people live in the region, of which more than 90% are Tibetan population and 68.15% are herders. The urbanization rate is very low, and only two towns have a population of more than 20 thousands. The annual net income reached 2352 yuan (US $290.05) in 2012. Most people believe in Tibetan Buddhism. It is also the highest and most extensive protected wetland area in the world and contains alpine swamp meadows, dark coniferous forests, marsh wetlands, glaciers and other natural habitats that support unique and rare wildlife.

The Sanjiangyuan National Nature Reserve (SNNR), established in 2003, is the largest nature reserve in China. It has an area of 0.15 million km^2 and consists of 18 protected subareas, each of them was divided into a core zone (C), an experimental zone (E) and a buffer zone (B) according to principle of the Nature Reserve (see Fig. 1). The 18 protected subareas were distributed in the three source regions of YR, HR and LM respectively (Table 1). In addition, the non-PAs of the three source regions were defined in each regions. The glaciers, permafrost and snows are widespread here, owing to its average altitude of over 4000 m and annual mean temperature of −5.6 to −3.8 °C. Due to climate change and human activities like overgrazing and mining, the water conservation functions provided by this region have been weakened, and grassland degradation, desertification, and soil erosion by rodents have been aggravated (PGQP, 2005; Liu et al., 2008; Dong et al., 2013). Habitat fragmentation also resulted in the loss of biodiversity. Since 2005, approximately $7.5 billion yuan (US $924.79 million) was invested to conserve and restore ecosystems in the SNNR (PGQP, 2005). In which, 65.6% was invested in environmental conservation and restoration, 29.6% was used to improve the resource production and living conditions of the local communities, and 4.8% was used for artificial rainfall, monitoring and research.

3. Material and methods

To monitor habitat quality and ecosystem dynamics in the SNNR, we performed a series of studies including field investigation, model simulation, remote sensing and GIS analysis to assess the ecological indices (Table 2).

3.1. Field investigation and observation

In the field, 280 ecosystem representative field plots in the SNNR were investigated to collect data on ecosystem change. Of these, 113 were located in grasslands, 120 in forests, 23 in wetlands, and 24 in deserts. In grassland plots, five 1 × 1 m quadrats were established in a 50 × 50 m region, and community-level investigation for vegetation coverage, below-ground and above-ground biomass, and soil water content were observed annually. In forest plots, forest coverage, diameter at 1.3-m height, tree height, and stand volume were measured in randomly placed 30-m^2 plots.

We collected daily observation data on precipitation and air temperature from 1990 to 2012 at 17 national meteorological stations to analyze the variability of climate conditions. The daily run-off data from 1990 to 2012 at four main hydrological observation stations were also collected to monitor changes in run-off.

3.2. Remote sensing interpretation and detection

Satellite-based approaches offer an inexpensive and verifiable way to quickly design and apply adaptive management strategies for PAs (Pettorelli et al., 2012). We performed remote sensing analysis by using Landsat TM/ETM+ and ArcGIS platforms to survey changes in ecosystem type and to provide information on ecosystem changes (Liu et al., 2008; Xu et al., 2008). The images were acquired primarily in July and August. False color compositions (Landsat bands 4, 3, 2), geometric corrections, mosaic images and segmentations were processed using base map data and ground control points. Land use and land cover datasets were produced for 1990, 2004 and 2012 and then classified into ecosystem types consisting of forests, grasslands, wetlands, deserts and others.

To reflect the dynamics of grassland ecosystems more accurately, especially when degraded and restored, we classified the changing trend of grasslands during 2004–2012 based on the former status of grassland degradation in the period of 1990–2004. The changing situation of degraded grasslands from 2004 to 2012 was estimated by comparing TM/ETM+ images from 2004 and 2012, the difference in which was classified as new occurring degradation (ND), intensified degradation (ID), slight restoration (SR), or obvious restoration (OR). The selected criteria to define these classes were referred to Liu et al. (2008) and Xu et al. (2017).

The 1 km SPOT-VGT NDVI for 10-day intervals from 1998 to 2012 was collected, and the annual maximum NDVI values through a maximum value composite were then produced. The 1 km maximum NDVI values were applied to calculate the annual maximum vegetation coverage using the dimidiate pixel model (Leprieur et al., 1994),

$$f = (NDVI - NDVI_{min})/(NDVI_{max} - NDVI_{min}) \tag{1}$$

where f is fractional vegetation coverage, $NDVI_{min}$ is the NDVI value for bare land without vegetation, and $NDVI_{max}$ is the NDVI value of pure vegetation. The bare land and pure vegetation were selected according to land use and land cover datasets, and average value were estimated. The calculated fractional vegetation coverage was validated using data from measured field plots. The vegetation coverage value was then analyzed and compared between the periods of 1998–2004 and 2004–2012 to determine whether vegetation had been restored or had degraded.