

Dynamic changes of habitats in China's typical national nature reserves on spatial and temporal scales

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Abstract: Until 2015, China had established 2740 nature reserves with a total area of 1.47 million km², covering 14.8% of China's terrestrial land surface. Based on remote sensing inversion, ecological model simulation and spatial analysis methods, we analyzed the spatial and temporal variations of fractional vegetation coverage (FVC), net primary production (NPP), and human disturbance (HD) in habitats of typical national nature reserves (NNRs) during the first 15 years of the 21st century from 2000 to 2015. And then the three indicators were compared between different NNR types and varied climate zones. The results showed that (1) the average 5-year FVC of NNRs increased from 36.3% to 37.1%, and it improved in all types of NNRs to some extent. The annual average FVC increased by 0.11%, 0.84%, 0.21%, 0.09%, 0.11% and 0.08% in NNRs of forest ecosystem, plain meadow, inland wetland, desert ecosystem, wild animal and wild plant, respectively. (2) The NPP annually increased by 2.06 g·m⁻², 1.23 g·m⁻², 0.28 g·m⁻² and 0.4 g·m⁻² in NNRs of plain meadow, inland wetland, desert ecosystem and wild animal, respectively. However, it decreased by 3.45 g·m⁻² and 2.35 g·m⁻² in NNRs of forest ecosystem and wild plant respectively. (3) In the past 15 years, besides the slight decreases in the NNRs located at the Qinghai-Tibet Plateau and the south subtropical zone, HD enhanced in most of NNRs, especially HD in the warm temperate humid zone increased from 4.7% to 5.35%.

Keywords: national nature reserves; habitat; fractional vegetation coverage; net primary production; human disturbance

1 Introduction

“Nature reserve” refers to certain areas (land, waters or sea area) with special legal protec-

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tion and management that aims to protect the representative nature ecosystem, rare and endangered wildlife species with natural concentrated distribution, and natural relics with special significance. Establishment of protected areas (PAs) is one of the most important approaches for biodiversity conservation (Howard *et al.*, 2000; MEA, 2003; Radeloff *et al.*, 2010; Stein *et al.*, 2008) and is a cultural response to the perceived threats to nature (McNeely, 1994). PAs are recognized as the most important core units for in situ conservation (Maiorano *et al.*, 2008). The value of nature reserves and what they are established to conserve are changing because society is constantly changing (Chape *et al.*, 2005). The International Union for Conservation of Nature (IUCN) classified PAs into nature reserves, national parks, natural monuments, habitat/species management areas, protected landscape/seascapes and managed resource protected areas (Anon, 1994; Wang *et al.*, 2004). Different types of PAs offer different levels of protection. China has their own classification of PAs, which is quite different from the IUCN management categories. In China, nature reserves are the most strictly managed type of PAs, and are classified into natural ecosystems (forest ecosystem, plain meadow, inland wetland, desert ecosystem), wildlife (wild animal, wild plant) and natural monuments (geological relics, archaeological remains), based on the main objectives of protection and the management goal to a certain extent (Xue *et al.*, 1994).

With increasing human pressure on the natural resources, effective PAs system is of great significance for conserving biodiversity or slowing the rate of biodiversity loss (Chape *et al.*, 2005). Held in Bali Island in 1982, the World Parks Congress recommended that all nations should strive to cover 10% of its terrestrial lands under protection (Naughton-Treves *et al.*, 2005). As of 2014, more than 0.2 million PAs had been established worldwide, which accounted for approximately 15.4% of the terrestrial area of the earth (Juffe-Bignoli *et al.*, 2014). The Ministry of Environmental Protection of the People's Republic of China announced that China had established 2740 nature reserves with a total area of 1.47 million km² as of 2015, covering 14.8% of China's terrestrial land surface and including 428 NNRs of all nature reserves. With the growth of the area under protection, research on PAs has become more significant (Soutullo, 2010). In recent years, some debates related to the effectiveness of PAs were discussed between academia and management departments (Liu *et al.*, 2003; Quan *et al.*, 2010; Thomas *et al.*, 2012). The importance of PAs under the background of climate change became the academic hotspot, because climate change may shift the distribution of species (Liu *et al.*, 2003).

Previous studies of nature reserves mainly focused on the setting rationality, the management effectiveness, the influencing factors of the nature reserves, the impact of nature reserves on the regional economy and society. However, due to the data qualities of species, the spatial precision of nature reserves' boundaries and other factors, the controversy occurred regarding the relative importance as to which areas need priority conservation. Under the background of climate change, the benefits of PAs for biodiversity have been questioned, because PAs are static, whereas the distributions of species are dynamic (Thomas *et al.*, 2012). The effectiveness of nature reserves depends on many local factors such as politics and economy, and there are many effectiveness evaluation methods (Hockings, 2003; Maiorano *et al.*, 2008). However, a unified standard has not yet been formed (Chape *et al.*, 2005). On the one hand, the effectiveness of PAs is increasingly threatened by climate change and human activities, although the degree of this threat is unknown. Urbanization, real estate

development and road construction seriously affected the nature reserves as a “Noah’s ark” protection and reduced the effective size of the nature reserves (Foley *et al.*, 2005; McDonald *et al.*, 2008). Global degradation of biodiversity is closely tied to the current and future urbanization (McDonald *et al.*, 2008). On the other hand, PAs have impact on the ecosystem by maintaining traditional livelihoods, keeping energy balance, regulating local climate, preventing forest fires, etc. (Walker *et al.*, 2009).

Most nature reserves are located in high-poverty areas. There is a contradiction between achieving ecosystem conservation goals by limiting resource exploitation and reducing or eliminating poverty (Naughton-Treves *et al.*, 2005). In developing countries, limiting the exploitation of natural resources may generate new poverty or reinforce existing poverty. Research also found that the establishment of nature reserves can reduce poverty because there is no correlation between poverty and reduced deforestation (Ferraro *et al.*, 2011). Some studies evaluated the effectiveness of reserves by comparing the inside and outside of PAs (Bruner *et al.*, 2001; Naughton-Treves *et al.*, 2005; Oliveira *et al.*, 2007), especially in rich, biodiverse tropical forest regions, such as the Amazon and the Congo (Joppa *et al.*, 2008; Walker *et al.*, 2009). Combined with different resolution levels of satellite images and field survey data, the effectiveness of nature reserves was evaluated by forest decrease or deforestation reduction (Radeloff *et al.*, 2010; Ferraro *et al.*, 2011). Intensive human activities were found in adjacent areas of nature reserves became hotspot areas (Barbier *et al.*, 2001; Andam *et al.*, 2008; Foley *et al.*, 2005). A large amount of forest set aside as a “green barrier” in the Brazilian Amazon region to eliminate spillover effects (Soares-Filho *et al.*, 2010).

Although the rapidly increasing quantity of nature reserves, many doubts have arisen that some reserves may be “paper parks” rather than achieving sustainable conservation outcomes (Liu *et al.*, 2003; Quan *et al.*, 2010). In China, the past research studying the rationality of the reserve setting, the effectiveness of reserve management, the influencing factors of reserves, and the regional social and economic impacts of reserves mainly concentrated on a single area or the individual index (Fan *et al.*, 2012; Zhang *et al.*, 2016; Zheng *et al.*, 2012). The incomplete datasets of biodiversity and habitat, the unclear boundaries of the nature reserves and the varied background of the ecosystem led to difficulties in evaluating the effectiveness of the nature reserves and understanding whether the nature reserves are achieving biodiversity conservation. In this paper, NPP and FVC were selected to respond to the condition and trends of ecosystem, and the dynamic changes of habitats and human disturbance (HD) were analyzed to illustrate the intensity of human activities. We chose typical NNRs as study areas. The spatial and temporal patterns of FVC, NPP and HD were analyzed, and dynamic changes of habitats in different NNR types and located in varied climate zones were evaluated.

2 Materials and methods

2.1 Study areas

In addition to sea coast and aquatic animal NNRs, we selected 299 typical NNRs established after the year 2000 as the study area. The typical NNRs consist of 167 forest ecosystem NNRs, 4 plain meadow NNRs, 36 inland wetland NNRs, 13 desert ecosystem NNRs, 67

wild plant NNRs, and 13 wild animal NNRs. The boundaries of the NNRs were collected from the website of the Ministry of Environmental Protection (<http://www.zhb.gov.cn/stbh/zrbhq/gjjzrbhqps/>) and digitalized by referring to topography and high-resolution satellite images. Terrestrial China was divided into seven climate zones according to China's new climatic scheme, proposed by Zheng *et al.* (2010), and combined with the regional geographic differences and the spatial distribution of the NNRs. Among them, 22 NNRs were located in the south subtropical zone, 102 NNRs in the north subtropical zone, 42 NNRs in the warm temperate humid zone, 38 NNRs in the Qinghai-Tibet Plateau, 42 NNRs in the middle temperate humid zone, 35 NNRs in the middle temperate semi-arid zone, and 19 NNRs in the middle temperate arid zone (Figure 1).

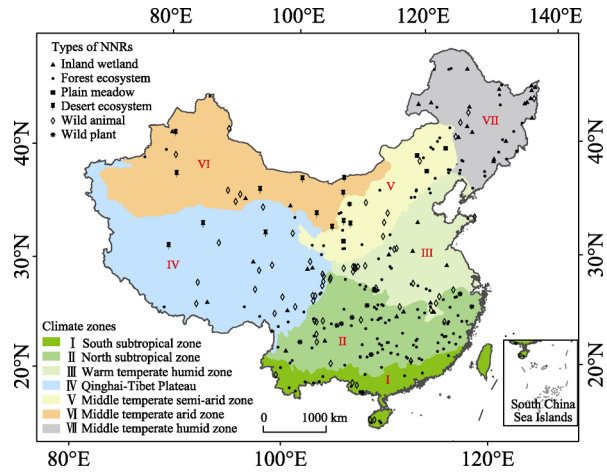


Figure 1 Spatial distribution of 299 typical NNRs in China

2.2 Estimation of fractional vegetation coverage

The MODIS NDVI products from 2000 to 2015 with 1 km spatial resolution and 16-day temporal resolution were collected and processed by S-G filtering methods. The fractional vegetation coverage was calculated by using the dimidiate pixel model based on the assumption that a pixel of NDVI mixed the information of green vegetation and non-vegetation. Therefore, FVC could be calculated by the following equation,

$$FVC = \frac{NDVI - NDVI_{\min}}{NDVI_{\max} - NDVI_{\min}} \quad (1)$$

where FVC is calculated with a 5% confidence interval. $NDVI_{\max}$ is the NDVI value of pure green vegetation pixels, and $NDVI_{\min}$ is the NDVI value of pure non-vegetation pixels.

The general change trends (slo) were analyzed by the least squares method as follows,

$$slo = \frac{n \times \sum_{i=1}^n (i \times FVC) - \sum_{i=1}^n i \sum_{i=1}^n FVC}{n \times \sum_{i=1}^n i^2 - \left(\sum_{i=1}^n i \right)^2} \quad (2)$$

where i is the serial number from the year of 2000 to 2015, $i=1, 2, 3, \dots, n$. If the slo is positive, the FVC change trend of that pixel is increased. Conversely, the FVC is decreased.

2.3 The model simulation of net primary production

The ecological model applied to simulate NPP can be classified into statistical model, processing model, mechanism model and remote sensing model, with typical representatives such as the Carnegie Ames Stanford Approach (CASA), Global Production Efficiency Mod-

el (GLOPEM), Photosynthesis (PSN) and Vegetation Photosynthesis Model (VPM). The remote sensing model has the advantages of less input parameters, and the driving variables can be obtained from remote sensing data directly. Due to the uncertainties of model simulation, accurate simulations need various kinds of full evaluation by flux observation data at the ecosystem scale. GLOPEM was established based on the theory of physiological ecology, and it had been evaluated to achieve ideal outputs based on the validation of the forest ecosystem and the plain meadow after parameter localization (Wang *et al.*, 2009).

In this study, the GLOPEM model was applied to simulate NPP, which is calculated by the difference between gross primary production (GPP) and autotrophic respiration (R_a).

$$NPP = GPP - R_a \quad (3)$$

GPP is calculated by modeling efficiency for solar energy utilization (ε_g), fraction of photosynthetically active radiation (FPAR), and modeling absorbed photosynthetically active radiation (APAR).

$$GPP = APAR \cdot \varepsilon_g \quad (4)$$

$$APAR = PAR \cdot FPAR \quad (5)$$

where PAR is photosynthetically active radiation, determined by the radiation calculation method in climatology.

R_a was calculated as follows,

$$R_a = f(R_m) + R_g \quad (6)$$

where R_m is the maintenance respiration, and R_g is the growing respiration.

Finally, NPP is simulated from 2000 to 2015 with a spatial resolution of 1 km for each typical NNR, and the NPP change trends are analyzed by the least squares method.

2.4 Quantification of human activities

The land use and land cover datasets for the years 2000, 2005, 2010 and 2015 were collected from Liu *et al.* (2014) with a 100 m spatial resolution, including 6 aggregated classes and 25 hierarchical classes. The datasets were evaluated by using the field investigation of Zhang *et al.* (2012). According to evaluation verification in 10% of the counties in China, the comprehensive evaluation accuracy was 94.3% (Liu *et al.*, 2014).

Based on the land use and land cover datasets, human disturbance is defined as the degree of utilization, transformation, and exploitation of the land surface in a certain region (Xu *et al.*, 2015). We analyzed the area changes of artificial land use types (cropland, built-up areas) in NNRs, and then calculated human disturbance with a spatial resolution of 100 m. Human disturbance is calculated by the following equations,

$$HD = \frac{S_c}{S} \times 100\% \quad (7)$$

$$S_c = \sum_{i=1}^n SL_i \cdot CI_i \quad (8)$$

where HD is an index of human disturbance; S_c is the equivalent area of construction; S is total area of study area; i is the type of land use and land cover change; SL_i is the real area of i ; CI_i is a conversion factor for land use and land cover transformed to built-up areas, according to the strength of the human activities on the land surface.

Based on the attributes of the land surface, the values of CI_i (Table 1) were defined by re-

ferring to Xu *et al.* (2015).

Table 1 Conversion factor of different land use and land cover types

Land use and land cover	Cropland	Artificial forest	Reservoir	Built-up areas	Others
Conversion factor	0.2	0.133	0.6	1	0

3 Results and analysis

3.1 The spatial and temporal variations of FVC in NNRs

From 2000 to 2015, the annual average FVC of NNRs increased by 0.8% from 36.3% to 37.1%, with an annual growth of 0.13% for the 15 years (Figure 2a). Of all types of NNRs, the FVC in forest ecosystem NNRs increased by 0.5% from 69.4% to 69.9%, with an annual growth of 0.11%. FVC in plain meadow NNRs increased by 5.6% from 59.8% to 65.4%, with an annual growth of 0.84%. In inland wetland NNRs, FVC increased by 1.5% from 58.9% to 60.4%, with an annual growth of 0.21%. In desert ecosystem NNRs, it increased by 0.5% from 12.8% to 13.3%, with an annual growth of 0.09%. FVC in wild animal NNRs increased by 0.7% from 32.6% to 33.3%, with an annual growth of 0.11%. Conversely, FVC in wild plant NNRs decreased by 0.2% from 47.2% to 47% (Figure 2a).

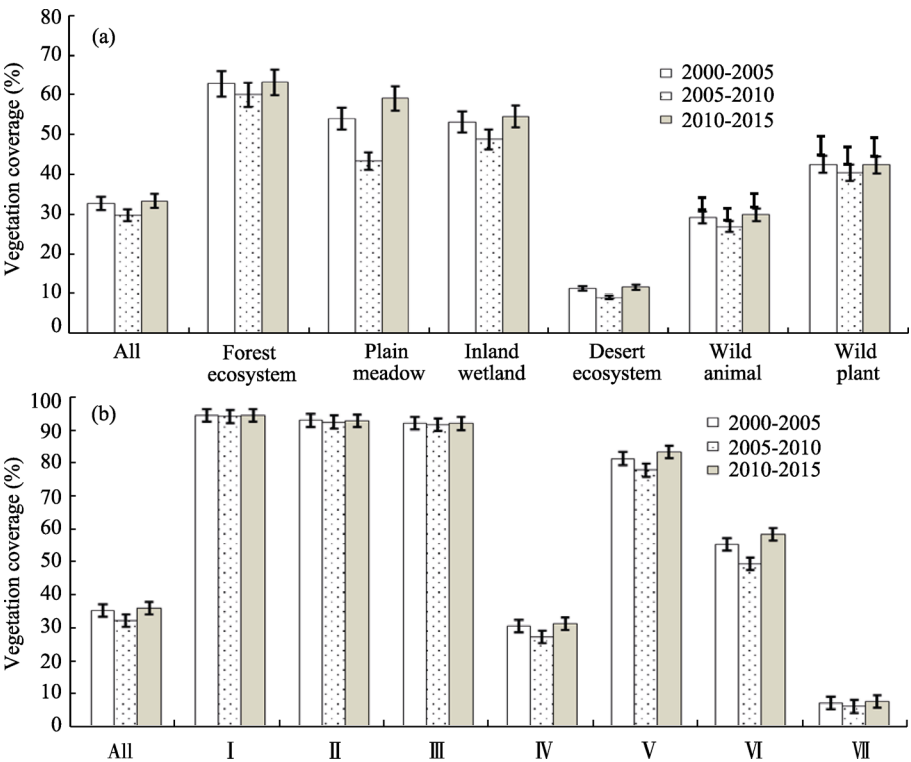


Figure 2 The average FVC in typical NNRs (a) with different types and (b) located in varied climate zones

Of all climate zones (Figure 2b), the FVC of NNRs in the middle temperate humid zone increased by 2.1%, from 83.9% to 86%, with an annual growth of 0.27%. The FVC of NNRs in the middle temperate semi-arid zone increased by 3.2%, from 57% to 60.2%, with an an-

nual growth of 0.51%. The FVC of NNRs in the middle temperate arid zone increased by 0.5%, from 7.3% to 7.8%, with an annual growth of 0.07%. The FVC of NNRs in the Qinghai-Tibet Plateau increased by 0.6%, from 31.5% to 32.1%, with an annual growth of 0.12%. The FVC of NNRs in south subtropical zone increased by only 0.1%. The FVC of NNRs in north subtropical zone decreased from 95.9% to 95.8% and has maintained 94.9% in warm temperate humid zone.

From the spatial variations of FVC in typical NNRs from 2000 to 2015 (Figure 3), we can see that the increasing trends occurred in the northern Qinghai-Tibet Plateau, middle temperate arid and semi-arid zones, and parts of the middle temperate humid zone. Especially, the increasing trends of FVC were obvious in the NNRs of Sanjiangyuan, Qiangtang, Mount Qomolangma, Hoh Xil, Altun Mountain, Qilian Mountain, Lop Nor, Xilingol League, Erdos, and Hongze Lake Wetland. The annual average FVC showed slight increases in NNRs of Greater Khingan Range, West Lake of Dunhuang, Zoige Wetland, Haizi Mountain, and Aden. However, FVC evidently decreased in NNRs of Selin Co, the middle reaches of the Yarlung Zangbo River, Wolong, Taibai Mountain, Eastern Dongting Lake, Nanling Mountains, Wuyi Mountains, and Changbai Mountain.

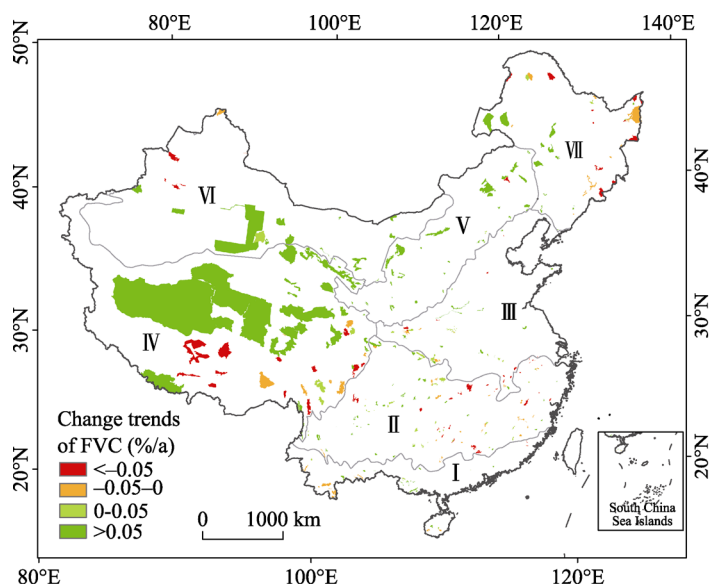


Figure 3 Spatial variations of FVC in typical NNRs from 2000 to 2015

3.2 The spatial and temporal variations of NPP in NNRs

From 2000 to 2015, the annual average NPP of NNRs decreased from $140.49 \text{ g} \cdot \text{m}^{-2}$ to $139.77 \text{ g} \cdot \text{m}^{-2}$ with an annual decline of $0.04 \text{ g} \cdot \text{m}^{-2} \cdot \text{a}^{-1}$ for the 15 years (Figure 4a). Of all types of NNRs, NPP in forest ecosystem NNRs decreased by $35.9 \text{ g} \cdot \text{m}^{-2}$, from $398.2 \text{ g} \cdot \text{m}^{-2}$ to $362.3 \text{ g} \cdot \text{m}^{-2}$, with an annual decline of $3.45 \text{ g} \cdot \text{m}^{-2} \cdot \text{a}^{-1}$. NPP in wild plant NNRs decreased by $28.3 \text{ g} \cdot \text{m}^{-2}$, from $277.2 \text{ g} \cdot \text{m}^{-2}$ to $248.9 \text{ g} \cdot \text{m}^{-2}$, with an annual decline of $2.35 \text{ g} \cdot \text{m}^{-2} \cdot \text{a}^{-1}$. Meanwhile, NPP in plain meadow, inland wetland, desert ecosystem, and wild animal NNRs increased by $15.4 \text{ g} \cdot \text{m}^{-2}$, $14.1 \text{ g} \cdot \text{m}^{-2}$, $1.9 \text{ g} \cdot \text{m}^{-2}$, and $1.4 \text{ g} \cdot \text{m}^{-2}$, respectively.

From the spatial variations of NPP in NNRs from 2000 to 2015 (Figure 5), we can see that increasing trends occurred in the Qinghai-Tibet Plateau, middle temperate semi-arid

zone and partly middle temperate humid zone. Especially, the increasing trends of NPP were obvious in NNRs of Zoige Wetland, Haizi Mountain, Aden, Zhouzhi, Taibai Mountain, Daqing Mountain, Xilingol League, and Wutai Mountain. The annual average NPP showed a slight increase in NNRs of Sanjiangyuan, Qiangtang, Mount Qomolangma, Hoh Xil, Altun Mountains, Qilian Mountain, Selin Co, Gongga Mountain, West Erdos, Helan Mountains in Inner Mongolia, Horqin, and the Greater Khingan Range. However, NPP evidently decreased in NNRs of Yarlung Zangbo Grand Canyon, Xishuangbanna, Leigong Mountain, Nanling Mountains, Wuyi Mountains, Changbai Mountain, and the Songhua River.

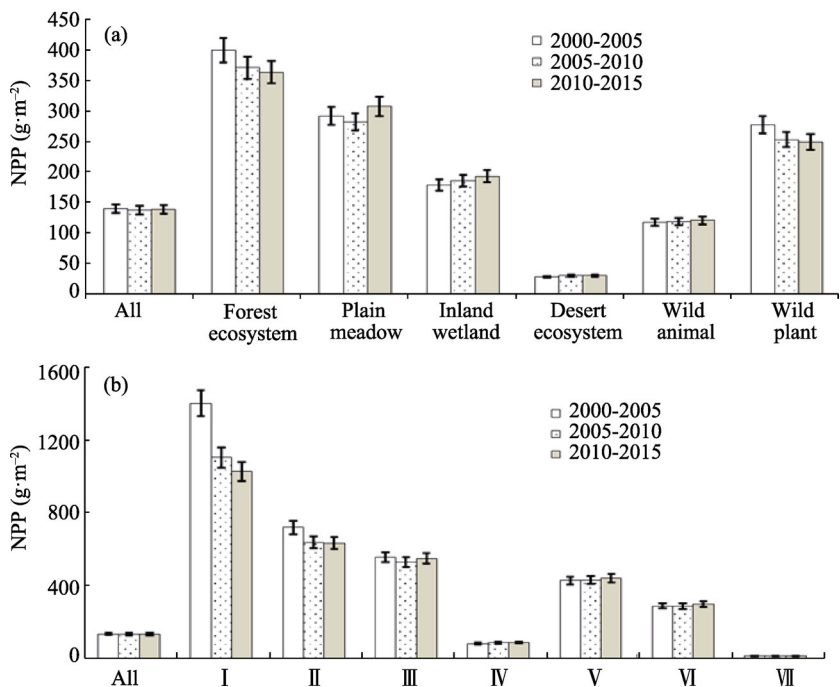


Figure 4 The average NPP in typical NNRs (a) with different types and (b) located in varied climate zones

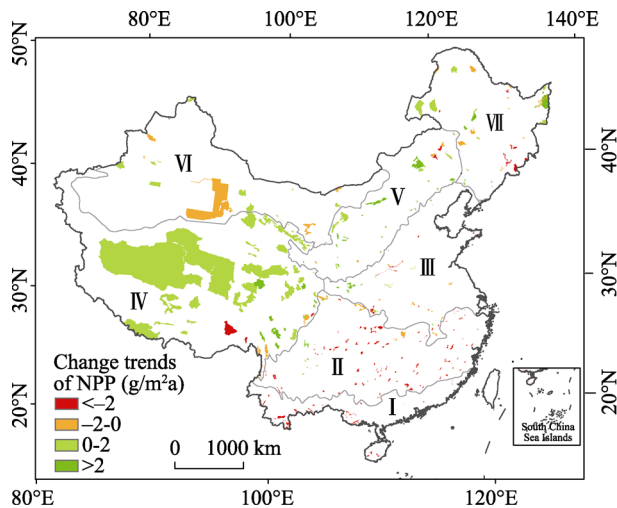


Figure 5 Spatial variations of NPP in typical NNRs from 2000 to 2015

3.3 Dynamics of human activities in the past 15 years

From 2000 to 2015, the area of cropland increased in 44 typical NNRs and decreased in 125 typical NNRs. The built-up areas increased in 111 typical NNRs and decreased in 15 typical NNRs. In different types of NNRs, we can see that the area of cropland increased the most in inland wetland NNRs (Figure 6a), a response to the enhanced disturbance of agricultural activities. The decreased cropland area in forest ecosystem NNRs means that agricultural activities were well limited. The built-up areas increased the most in the middle temperate semi-arid zone and the least in the middle temperate humid zone (Figure 6b).

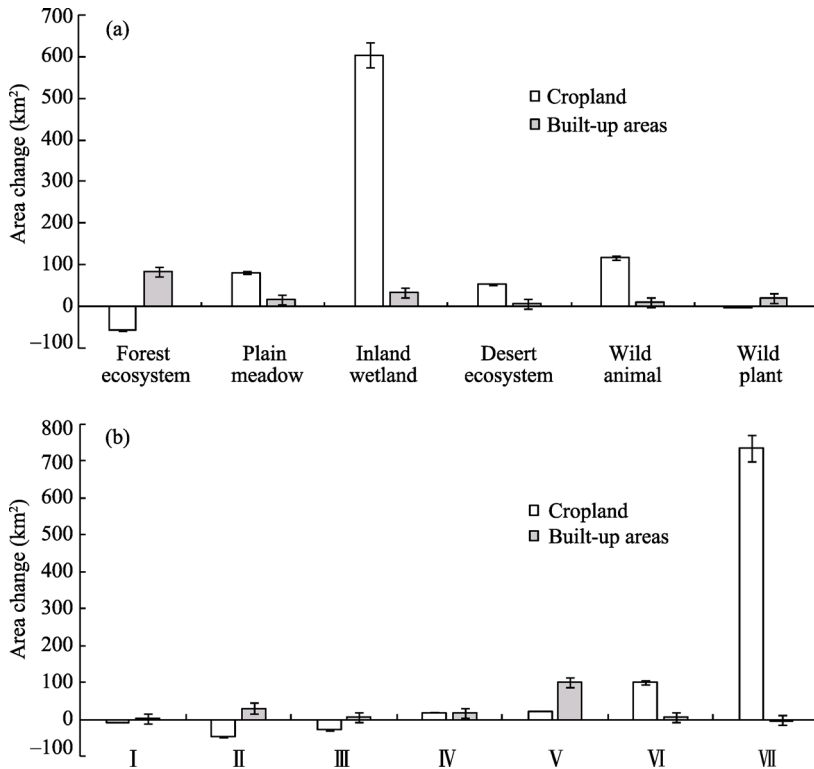


Figure 6 The area changes of cropland and built-up areas in typical NNRs (a) with different types and (b) located in varied climate zones

The average HD in typical NNRs decreased from 5.72% to 4.83% in the past 15 years. Of all types of NNRs, HD decreased from 6.93% to 4.49% in desert ecosystem NNRs and decreased from 4.04% to 3.47% in wild animal NNRs. In contrast, HD increased from 3.61% to 3.93% in forest ecosystem NNRs, increased from 7.22% to 7.44% in plain meadow NNRs, dramatically increased from 6.69% to 7.37% in inland wetland NNRs, and increased from 5.5% to 6.02% in wild plant NNRs.

In all climate zones, HD dramatically decreased from 6.46% to 5.32% in NNRs of the Qinghai-Tibet Plateau, especially in the NNRs of Qiangtang, Altun Mountains, Selin Co, and Haizi Mountain. However, the slight increasing trend of HD in the entire Qinghai-Tibet Plateau (Xu *et al.*, 2015; Zhao *et al.*, 2015) showed the effectiveness of NNRs. HD in the NNRs of the middle temperate arid zone decreased dramatically from 2.16% to 1.44%, which was especially obvious in the NNR of the Tarim Populus Euphratica. The obvious

decreasing trend of HD in the middle temperate arid zone may be affected by the Grain for Green Program (Liu *et al.*, 2014; Xu *et al.*, 2015; Zhao *et al.*, 2015). Conversely, HD evidently increased in typical NNRs located in other zones. The HD of NNRs increased from 1.54% to 2.06% in the south subtropical zone, from 2.99% to 3.24% in the north subtropical zone, from 4.7% to 5.35% in the warm temperate humid zone, from 4.61% to 4.93% in the middle temperate humid zone, and from 6.31% to 6.63% in the middle temperate semi-arid zone. In eastern China, HD occurred at a higher level and strengthened in recent years due to high population density and urbanization (Xu *et al.*, 2015; Zhao *et al.*, 2015). By contrast, the HD of NNRs in those regions showed a lower level and enhanced slightly (Figure 7).

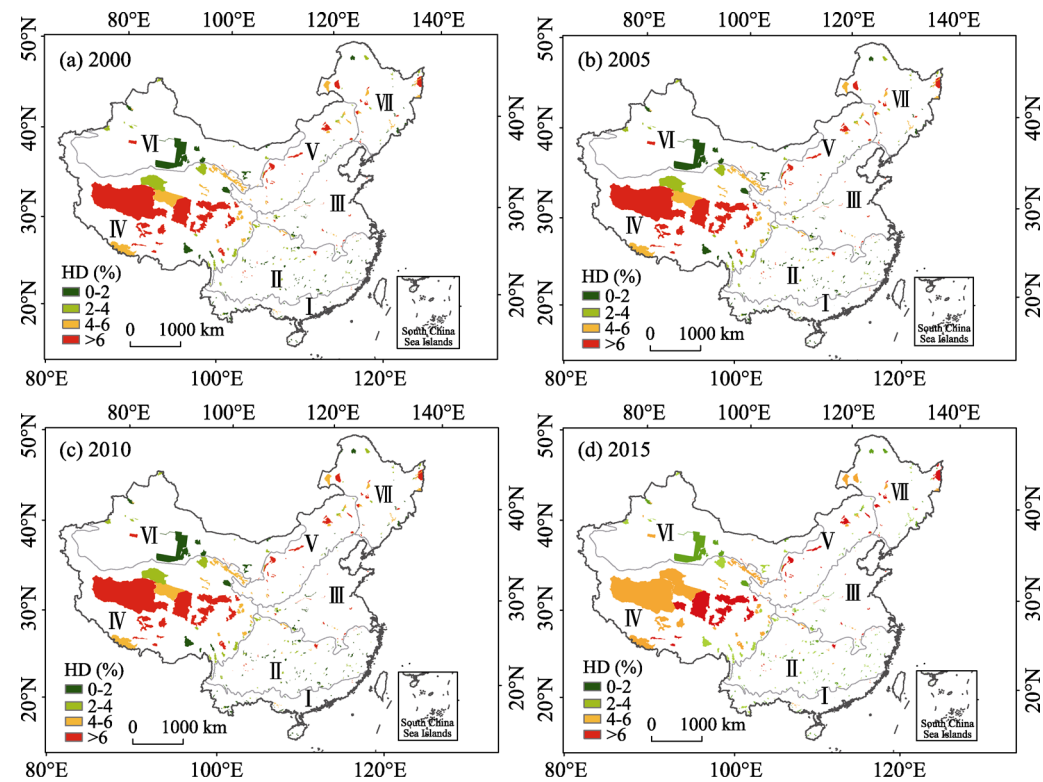


Figure 7 Spatial distribution of HD in typical NNRs for the years of 2000, 2005, 2010 and 2015

4 Conclusions and discussion

From 2000 to 2015, FVC of typical NNRs with different types and located in varied climate zones have an increasing trend generally. In addition to decreased NPP in forest ecosystem and wild plant NNRs due to deforestation, NPP in other types of NNRs increased in recent 15 years. The weakening trends of HD in typical NNRs in recent 15 years could be partly contributed to the effectiveness of limiting human activities in NNRs, especially in the middle temperate arid zone, the Qinghai-Tibet Plateau and parts of the middle temperate humid zone. However, increasing HD occurred in NNRs located in the North China Plain, hilly area of East China, and Loess Plateau, due to urban expansion and increasing population. Located in the East Asian monsoon region, China responds to global climate change sensitively. Climate change has huge impacts on ecosystems and is expected to be the main fac-

tors of species extinction in the 21st century (Thomas *et al.*, 2004; Pereira *et al.*, 2010). The impact of climate change on ecosystems has evoked a lot of discussion in recent years. Studies have shown that the impacts of climate change on species are mainly reflected in the composition and interaction of population, the range and distribution of the species, NPP, and the structure and changes of ecosystems (Beaumont *et al.*, 2011). Both climate change and human activity play important roles in biodiversity and ecosystem services (Pereira *et al.*, 2012; Titeux *et al.*, 2016; Struebig *et al.*, 2015). Many studies on the impact of climate change on biodiversity (Bellard *et al.*, 2012; Staudinger *et al.*, 2013; Pacifici *et al.*, 2015) showed that future climate change is predicted to generate latitudinal or altitudinal shifts in species ranges (Maes *et al.*, 2010; Barbet-Massin *et al.*, 2015), reduce the effectiveness of conservation areas (Araújo *et al.*, 2011), or increase the risks of species extinction (Thomas *et al.*, 2004; Urban, 2015).

In the Qinghai-Tibet Plateau, climate change directly influences the starting date of the vegetation growing season, and it may expand the length of the plant growing season, which is closely related to NPP (Zhang *et al.*, 2013; Shen *et al.*, 2015). The degradation of biodiversity and ecosystems is concurrently impacted by human activities. Studies showed that the interior of some nature reserves are still under the influence of human activities, especially in coastal areas, with the growth of population density and the expansion of urbanization. Human activities in NNRs mainly include the construction of roads and other facilities, and unreasonable exploitation activities such as rock excavation, sand excavation and mining. As the most strictly managed types of PAs, nature reserves should have limited or even forbidden human activities within their areas.

Although the protection from human activities continues to increase, the habitats of NNRs are influenced both by climate change and human activities, and it is difficult for us to judge the contribution of NNRs to the habitat changes. Distinguishing the contribution rate of climate change and human activities will be the key step toward assessing the effectiveness of PAs and will be also the further research focus. Our research is incomplete, because it used only three indicators of FVC, NPP and HD to evaluate the habitat changes of nature reserves. Different types of nature reserves have different protection purposes, it is insufficient to evaluate them with these few indicators. In view of the shortcomings, we should establish a corresponding index system for different types of nature reserves to assess habitat change properly, identify the function of nature reserves under the background of climate change, and evaluate habitat change and ecosystem vulnerability under future climate scenarios. In addition, we should further explore how to reduce the impacts of human activities and improve the effectiveness of NNRs under the background of rapid urbanization and increasing population. Furthermore, it is necessary to compare the conditions inside and outside of the NNRs.

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