

**EFFECTS OF CONSERVATION POLICIES ON FOREST COVER
CHANGE IN PANDA HABITAT REGIONS, CHINA**

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ABSTRACT

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To restore forests and protect remaining natural forests, in 1998 the Chinese government initiated two nationwide conservation policies, the Natural Forest Conservation Program (NFCP) and the Grain-To-Green Program (GTGP). This study evaluated the effects of conservation policies and other potential driving forces on forest-cover change in 108 townships located in the Qinling Mountains and Sichuan Giant Panda Sanctuary (both known giant panda habitat regions) between 2001 and 2008. Forest-cover change was evaluated using land-cover products derived from the Moderate Resolution Imaging Spectroradiometer (MODIS). Most townships in both regions showed either stable or increased forest cover. An Ordinary Least Square (OLS) regression model was applied to identify potential driving forces of this forest-cover change. The model suggests that conservation policies had significantly positive effects on forest cover, while population density, percentage of agricultural population, road density, and initial forest cover (i.e., in 2001) had significantly negative effects. This study helps to clarify not only the patterns of forest-cover change after conservation policy implementation, but also to identify the impacts of potential driving forces of forest-cover change, at township level. This information could be, in turn, useful in the development of future giant panda habitat restoration projects.

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INTRODUCTION

About 30 percent of world's land areas are covered by forests (FAO 2007). Forests provide valuable ecosystem services and support a high diversity of life-forms (Bawa and Dayanandan 1997). Changes in forest cover often result from human and natural forces operating at various spatio-temporal scales (Zhan et al. 2002; Hayes and Cohen 2007). Unprecedented rates of human population growth and other anthropological factors (e.g. timber harvest, cropland cultivation, infrastructure construction) have caused global land-cover changes in the past centuries, particularly from natural forests to other land cover types (Myers 1990; Pahari and Murai 1999; Carr 2004; Carr 2005). The shrinkage of natural forests around the world and the associated loss of plant and animal species and of the protective environmental functions provided by forests (Mather 1992) have been recognized as important environmental problems that humans are currently facing.

However, while the overall amount of forest cover has been declining worldwide, an opposite trend of forest-cover change (from shrinkage to expansion) has occurred since the late 18th century in some European countries. For example, historical documents show that the overall forest cover of France started to increase from 1789 after a significant reduction over thousands of years (Mather 1992). Forest expansion has also occurred in many other European countries, such as the Netherlands, Denmark, Switzerland, and Scotland since the 1800s. Furthermore, the developed world as a whole was at or close to a turn-around period such that deforestation was being halted and reforestation and afforestation were becoming more prevalent, particularly from the mid-1980s (Mather 1992). Japan, as the earliest industrialized country in

Asia, has experienced forest expansion since the late 18th Century, which is as early as many European countries (Totman 1986). South Korea started to exhibit reforestation in the 1960s (An et al. 2001). In North America, the reforestation trend first emerged in the New England region of the United States in the mid-19th Century (Foster et al. 1998). With the spread of industrialization and urbanization, the trend of increase in forest cover also appeared in many developing countries across the world. In Asia, four major developing countries -- China, India, Vietnam, and Bangladesh -- have been experiencing forest regeneration since the 1980s (Rudel 2005; Mather 2007). A similar trend of positive forest-cover change is also identified in Central and South American countries such as Mexico, Ecuador, and Brazil in recent decades (Klooster 2003; Baptista and Rudel 2006; Farley 2007).

This turning point of forest-cover change from negative to positive was termed ‘forest transition’ (Mather 1992; Mather et al. 1998; Mather et al. 1999; Mather and Fairbairn 2000; Mather 2004). Grainger (1995) later argued that forest transition should be considered as a critical point of a long-term trend, which actually comprises two fundamentally different processes of land-use allocation: one is the decline in forest area, and the other is the recovery in forest area after the transition (i.e. the turning point). These processes are termed as national land-use transition and forest-replenishment period, respectively (Grainger 1995).

Although forest transition has occurred in many places around the world, the causes of this phenomenon in different countries are highly variable. Three major reasons were summarized by Mather: “...expectations that current forest resources may not be adequate to meet future domestic wood consumption; changes in popular perceptions of forests, from areas to be exploited for timber or farmland, to resources deserving conservation for recreation or aesthetic qualities; and changes in land use, from abandonment of farmland as rural population decline or

farming intensifies on the most productive soil, allowing forest regeneration” (Mather 1992). Generally, forest transition in each country results from a specific combination of historical, political, and economic forces, but there may be some general patterns that repeat across the world during different historical periods (Rudel 1998). In order to explain the general processes of forest recovery after agricultural expansion, two lines of argument were brought out by Rudel and colleagues based on previous studies (Rudel 1998; Rudel et al. 2005). One argument is that deforestation raises the price of wood, which not only induces people to harvest the remaining primary forests but also encourages them to plant more trees (Sedjo and Clawson 1983). The argument is supported by much evidence around the world. For example, the scarcity of wood and related forest products has stimulated direct afforestation and reforestation by local villagers in West Africa, India, and the Philippines (Rush 1991; Fairhead and Leach 1995; Walters 1997). It pushes politicians to enact regulations and implement policies to increase forest cover as well (Haeuber 1993). Furthermore, higher prices of timber products also provide incentives for the private sector (e.g. farmland owners and timber companies) to plant their own forests in order to stabilize timber supplies (Prunty 1956; Hart 1968; Hart 1980; Royer 1987; Hardie and Parks 1996). The other prevalent argument states that industrialization results in a booming economy, which creates many off-farm job opportunities that attract laborers to migrate from agriculture to industry, which leads to the abandonment of marginal farmland and its conversion to reforested lands (Bentley 1989). In addition, advanced technology and the spread of industrialized agriculture has increased per-unit production and decreased the profit of farming marginal lands (Hart 1968). Meanwhile, a switch from fuelwood to fossil fuel during the urbanization process may further reduce human impact on forests (Mather 1990).

Although the ‘forest transition’ has been highly documented (Mather 1992; Grainger 1995;

Mather and Needle 1998; Rudel 1998; Rudel et al. 2005; Mather 2007; Barbier et al. 2010; Rudel et al. 2010) and an extensive body of literature exists on the many factors that play important roles as determinants of forest transition across the world (Kaimowitz 1997; Foster and Rosenzweig 2003; Klooster 2003; Perz and Skole 2003; Nagendra et al. 2005; Pan and Bilsborrow 2005; Lambin and Meyfroidt 2010), the previous binary rationale (i.e., wood scarcity and economic development) hardly explains all forest-transition phenomena. A variety of causal factors (driving forces) that operate under different environmental, socioeconomic, and political contexts are also important (Mather 2007). For instance, it has been found that development in some regions of Mexico is neither necessary nor sufficient to cause a forest transition, and reduced deforestation does not necessarily mean an eventual forest recovery (Klooster 2003). In addition, rapid secondary forest expansion does not always result from primary forest depletion (Klooster 2003; Perz and Skole 2003; Perz 2007). Therefore, it is important to develop a more thorough understanding of the driving forces behind forest transitions within various time spans (short-, medium-, and long run) and under various contexts.

Governments play important roles in facilitating forest transition. The loss of natural forests and the concomitant timber product scarcity, biodiversity loss, habitat fragmentation, and increase in the frequency of natural disasters (e.g., floods), bring both international and domestic pressures to national and local governments. They then react by establishing different mechanisms (e.g., policies) that try to preserve and/or restore forest cover (Grainger 1995; Mather 2007; Nagendra 2007). Therefore, the role of government policies should not be overlooked in forest transition theory, particularly in developing countries (Jack et al. 2008). As a part of government activities, Payments for Environmental (or Ecosystem) Services (PES) have emerged globally during the past few decades (Ferraro and Kiss 2002). These programs provide

incentives (direct and indirect) to individuals or communities for mitigating the overexploitation of natural resources and stopping the degradation of natural systems associated with them. Generally, direct incentives help implement conservation plans (e.g. wildlife habitat conservation, watershed protection, carbon sequestration) through land purchases, leases, and easements, which are based on a willing buyer-willing seller model. Indirect incentives are used to achieve similar results, through providing alternative economic and social benefits (e.g. forestry products and income through ecotourism) to enroll local individuals and communities (Ferraro and Kiss 2002). Although indirect incentives attract many stakeholders, since they focus on integrating conservation and development objectives based on the demands of nature conservation and human development, there is a growing recognition that indirect approaches have little impact because many environmental, political, and economic uncertainties (e.g. natural disasters and economic recession) may lower their cost-effectiveness (Ferraro 2001; Ferraro and Kiss 2002; Ferraro and Simpson 2002). Therefore, direct incentives have become prevalent, and more direct conservation payment programs have been initiated by governments and international non-governmental organizations around the world (Milne and Niesten 2009). These programs not only reward local communities for conservation activities, but also help them to develop alternative income opportunities (James et al. 1999; Ferraro 2001).

In addition to developed countries (e.g. United States, Australia, and several European countries) (Ferraro and Kiss 2002), some developing countries have also started implementing direct PES programs. In order to mitigate growing environmental problems, Costa Rica, as a pioneer of the developing world in dealing with deforestation by policy innovation, has introduced direct incentives for forest plantation since the 1970s (Pagiola 2008). In 1997, this country established the National Fund for Forest Financing and began implementing a

nationwide PES program to protect and restore forest systems, which can provide diversified environmental services (e.g. carbon sequestration, hydrological services, ecotourism and biodiversity conservation) (Chomitz et al. 1999; Castro et al. 2000). Through the PES program, providers of environmental services (e.g. farmers and forest owners) are paid for reforestation, forest conservation and sustainable forest management activities. As of 2005, about 10% of the country's forest area had been enrolled in this PES program (Pagiola 2008). Studies on PES in Costa Rica have found that deforestation trends have halted and PES recipients had higher forest cover than non-recipients after program implementation (Arroyo-Mora et al. 2005; Zbinden and Lee 2005; Pagiola 2008).

By providing incentives to willing buyers, PES schemes offer a direct, and possibly more equitable, method for achieving environmental outcomes than other approaches (Jack et al. 2008). However, the effectiveness of policy design and achievement of original goals are greatly impacted by the context in which a PES program is implemented. Thus, although many developing countries have been implementing PES projects during the past few decades, some outcomes obtained have been opposite to the intended goals. For instance, it has been reported that deforestation rates in areas that received PES payments are not necessarily lower than those observed in other areas without PES payments in Costa Rica (Rosa et al. 2003; Sanchez-Azofeifa et al. 2007). Therefore PES programs are not a panacea, since the expected outcomes may vary under various environmental, socioeconomic, and political contexts.

China is the largest developing country in the world. The demands of its large population and booming economy have caused deforestation and many other environmental problems, particularly during the last 60 years (Liu 2010). The drastic conversion from natural to human-dominated landscapes causes not only biodiversity loss but also fragmentation of wildlife

habitats (Loucks et al. 2001; Viña et al. 2007). Excessive timber harvest of natural forests and reclaiming farmland on hillsides of the upper reaches of the Yangtze and Yellow Rivers are considered the main reasons for the reduced water-retention capacity, increased siltation and floods (WWF 2003; Liu and Diamond 2005; Hu et al. 2006). Frequent droughts and floods during the 1990's in the Yangtze and Yellow rivers floodplain areas have demonstrated the urgency of stopping deforestation and expanding the areas under forest cover (WWF 2003). But it was only after suffering severe droughts in 1997 and huge floods in 1998 (Weyerhaeuser et al. 2005; Liu et al. 2008) that the Chinese government initiated two nationwide conservation programs [the Natural Forest Conservation Program (NFCP) and the Grain-to-Green Program (GTGP) in 1998 and 1999 respectively] to restore the degraded forest ecosystems. The main goals of the NFCP are to ban logging in the southwest and to reduce timber harvest in the northeast of China. Protection of natural forests in all regions is another important goal (Xu et al. 2006).

The main goal of the GTGP is to convert cropland to forests and grasslands in order to prevent soil erosion. Thus, one of the main criteria for selecting cropland to be included in the GTGP (Xu et al. 2006) is that slopes be $> 25^{\circ}$ (or $>15^{\circ}$ in some areas).

Government reports and documents declare that both conservation policies have thus far achieved the established goals. The targets of the NFCP for logging bans, harvest reductions, and forest protection have been achieved. By the end of 2008, the NFCP had protected about 108 million ha of natural forests and planted about 5.7 million ha with trees (SFA 2009b). Meanwhile, the GTGP has also achieved its goal in terms of tree planting. By the end of 2008, about 9.3 million ha of cropland in steep areas and 15.8 million ha of barren land have been planted with trees through the GTGP (SFA 2009c). Results of the 7th national forest resources survey (2004

through 2008) showed that forest cover in China has grown steadily since the previous survey, from 18.21% of the country's area by the end of 2003 to 20.36% by the end of 2008 (SFA 2009a).

Both conservation programs have drawn worldwide attention due to their operating scales, amount of public investments, and environmental implications, and several articles have been published on these programs (Xu et al. 2000; Zhao et al. 2000; Uchida et al. 2007; Wang et al. 2007; Xu et al. 2007; Liu et al. 2008). Some studies have focused on the ecological effects of these conservation policies. For example, a study in the northern part of Shaanxi Province indicated that the GTGP has contributed to an increase in vegetation cover between 1998 and 2005 (Cao et al. 2009). Chen and his team have concluded that the carbon stock of the Yunnan Province will increase during the next couple of decades if the GTGP continues (Chen et al. 2009). Therefore, the overall ecological effects of the NFCP and GTGP are generally perceived to be beneficial (Liu et al. 2008). Other researchers have focused on the socioeconomic perspectives. For instance, a study of socioeconomic benefits of the NFCP conducted by Shen and colleagues concluded that the NFCP will have positive economic impacts not only on the forestry sector but also on the economy of the entire China (Shen et al. 2006). However at the regional level, Cao and his team found that the NFCP is adversely impacting the livelihoods of local people in northwestern China due to the logging and grazing bans established through the NFCP (Cao et al. 2010). Compared with the NFCP, previous studies have paid more attention to the socioeconomic effects of the GTGP than on its ecological effects. From the socioeconomic perspective, implementation of the GTGP has produced mixed results.

On the one hand, Uchida and colleagues found that households participating in the GTGP not only obtained more income than non-participating households but also shifted their labor

from on-farm to off-farm (Uchida et al. 2007; Uchida et al. 2009). Other researchers have also reported similar results in which the livelihoods of enrolled farmer households seem to be better than those of non-enrolled households (Ye et al. 2003). However, several studies conducted in other regions have shown that the GTGP made little contribution to the income of participating households (Xu et al. 2004; Xu et al. 2007).

Most published studies have focused on the evaluation of social, economic, and ecological effects of these two conservation programs at either the macro (e.g., national) or micro (e.g., household) levels and in different study areas (Uchida et al. 2007; Xu et al. 2007; Liu et al. 2008). But very few studies have been conducted at intermediate levels (e.g. township and counties) (Trac et al. 2007; Zhou et al. 2007). The township level, in particular, is highly relevant because townships constitute the basic implementation unit of the NFCP and GTGP (Zhu and Feng 2003; www.gov.cn 2005). In addition, townships are the basal stratum of the overall 5-level planning system (i.e. National-Provincial-City-County-Township) for land use in China (OU et al. 2002). As a basal administrative level, township-level statistical data are systematically collected each year, which are not only an important data source of statistical yearbooks for higher administrative levels (e.g. county, province), but also provide relatively sufficient, proximate and accurate socioeconomic indicators that can be used for identifying driving forces of land-cover change. Township governments are in charge of making specific annual plans based on socioeconomic and biophysical conditions of the township, as well as tasks directly assigned by higher level governments (Zhu and Feng 2003). So each township may have different specific implementation schemes under the general regulation, which may produce various outcomes (e.g., differential changes in forest cover).

As budgets for conservation programs are usually limited, it is absolutely crucial to

evaluate the effectiveness of conservation programs in different contexts, which will guarantee scarce funds to go as far as possible in achieving conservation goals (James et al. 1999; Ferraro and Pattanayak 2006). Conservation policies, together with other driving forces (e.g. demographic, economic, technological, cultural and biophysical) may be the most important determinants of land-cover and land-use change (Turner et al. 1993; Geist and Lambin 2001). Research on driving forces of land-use and land-cover change will improve projections of land use in the future and comprehension of human responses to environmental change (Turner et al. 1993; Lambin et al. 2001). Therefore, in this study I attempt to answer the following three questions at the township level: (1) What are the patterns of forest-cover change since the implementation of conservation policies? (2) Do conservation policies have positive effects on forest cover? (3) Which underlying driving forces have significant effects on forest-cover change in the study area?

METHODS

Study Area

My study area is composed of two regions located in two different provinces. The first study region is located in the middle part of the Qinling Mountains, Shaanxi Province. It includes 57 townships in three counties (Zhouzhi, Foping, and Yang). The second study region is the UNESCO Giant Panda Sanctuary, located in Sichuan province, and includes 72 townships in twelve counties (Baoxing, Chongzhou, Dayi, Dujiangyan, Kangding, Li, Luding, Lushan, Qionglai, Tianquan, Wenchuan, and Xiaojin) (Figure 1).

Qinling Mountains

The Qinling Mountains are an important landmark in China. They not only constitute the natural boundary between southern and northern China, but also divide the Yangtze and Yellow River basins (Pan et al. 1988; Loucks et al. 2003). The elevation of the Qinling Mountains ranges from less than 1000 m to 3750 m (Figure 2). Many valleys and streams are located within these mountains. As a natural barrier, the Qinling Mountains block the cold, dry air coming from the north and maintain warm, humid air in the southern slope. Due to their special location, unique function, and diversified topographic characteristics, the Qinling Mountains are also a region with abundant biodiversity (SFA 2006) and home to many rare species, including the giant panda (*Ailuropoda melanoleuca*). Approximately twenty percent of all wild giant pandas (ca. 1,600 individuals) live in the Qinling Mountains (SFA 2006), distributed in three populations (Xinglongling, Taibaishan, and Niuweihe). In their entire geographic range, the wild giant pandas have the highest density in the Qinling Mountains region (Jiang 2006). As of 2009,

seventeen national and provincial nature reserves for the conservation of giant pandas have been established or are being planned in the Qinling Mountains (Xu et al. 2010). Besides the giant panda, many other rare species also live in the region, such as the Sichuan sub-nose monkey (*Rhinopithecus roxellanae*), the takin (*Budorcas taxicolor bedfordi*), and the red ibis (*Nipponia nippon*) (Pan et al. 1988). A total of 87 mammal species and 340 bird species have been recorded in the Qinling Mountains (Jiang 2006). In addition to animal species, the Qinling Mountains also have abundant plant species. In the southern slope alone, nearly 3,000 plant species have been recorded (Jiang 2006). The Qinling Mountains have been recognized as the one of the Global 200 Ecoregions defined by WWF (Olson et al. 2001).

The history of human activities in the Qinling Mountains can be traced to about 2,000 years ago. Zhouzhi County, which is located in the middle part of the Qinling Mountains and is also a part of my study region, was established in 104 B.C. (Jiang 2006). Large-scale timber harvest began about three hundred years ago, with hundreds of thousands of people depending on timber and related industries (Pan et al. 1988; Jiang 2006) for their sustenance. During this timber harvest period, in addition to the logging industry, the huge population also impacted the natural resources through cultivation, construction, and fuelwood collection for cooking and heating, all of which caused drastic forest-cover losses. Today, such encroachment on the forest still continues, but it is reported that it has been reduced through conservation policy implementation (SFA 2006). Modern commercial logging in this area started in 1957 and lasted till 1998, when the logging ban was implemented (Shaanxi 2002).

Sichuan Giant Panda Sanctuary

The Sichuan Giant Panda Sanctuary was established as a member of the UNESCO World Heritage System in August of 2006. It is home to more than thirty percent of the entire wild giant

panda population (SFA 2006). Seven nature reserves lie within the Sanctuary and aim to protect the giant panda and its habitat.

The elevation within the Sanctuary varies significantly (from ca. 400 to 6,200 m) (Figure 2). The entire region is dissected by valleys of perennial rivers flowing from snow-covered peaks and alpine meadows. The topographic characteristics and substantial variability of climates and soil types in the Sanctuary produce an enormous biological diversity containing between 5,000 and 6,000 plants species in more than 1,000 genera, with 50 genera being endemic to China (UNESCO 2006).

Besides plant species, the sanctuary also has diverse vertebrate species. It features 109 species of mammals and 365 species of birds. Like the Qinling Mountains, in addition to the giant panda, the Sanctuary is also the refuge of many other globally endangered mammal species, such as the red panda (*Ailurus fulgens*), the snow leopard (*Uncia uncia*), and the clouded leopard (*Neofelis nebulosa*), which make it a very important region for biodiversity conservation. In fact, the region has been classified as one of the world's top 25 Biodiversity Hotspots (Myers et al. 2000) and one of the Global 200 Ecoregions defined by WWF (Olson et al. 2001).

This region has been subject to human pressures for a long time. Besides the demands of local inhabitants, commercial timber harvest has occurred in this area since about four hundred years ago (Yang and Li 1992). From the mid-1950s to the late 1980s, due to growing demands of infrastructural and industrial development, intensive commercial logging was conducted across the Sichuan province. During that period, there were about 80 state-owned commercial logging enterprises and thousands of local lumber camps, which caused large-scale forest cover losses (Yang and Li 1992; SFA 2006). Before 1998, commercial logging completely depleted available forests in some areas (Wan 2009). After the conservation policies were implemented, all these previous commercial logging enterprises were closed or switched their functions from logging to planting trees and protecting forests (Luo et al. 2006).

For several reasons, both regions are ideal for evaluating the effects of conservation policies on forest cover. First, both areas suffered intensive commercial logging before 1998, when the national logging ban was implemented. During that period, different levels of commercial logging enterprises were operating across the entire study area, from large-scale state-owned forestry bureaus to small-scale collectively-owned logging farms (Pan et al. 1988; Yang and Li 1992). In 2000, the NFCP and GTGP started in both regions. As the basic policy implementation unit, each township is covered by one or both conservation policies, so it is possible to evaluate the effects of conservation policies on forest-cover change at the township level. Second, most townships are located in a mountainous region, with various biophysical attributes. The similarities and dissimilarities of these attributes may contribute to the changes in forest cover. Third, various socioeconomic characteristics (e.g. demographic, economic) may also cause different effects brought by similar conservation efforts. As most townships in the two study regions have systematically published township-level statistical data in multiple consecutive years, it is possible to obtain the necessary socioeconomic data at the township level. Finally, both regions include important giant panda habitat, so the study of forest cover change in the study region will provide information suitable for giant panda habitat conservation.

Hypotheses

Based on field observations and previous studies, I proposed the following hypotheses:

1. Forest cover in most townships either stabilized or increased after the implementation of conservation policies.
2. Demographic variables have negative effects on forest-cover change. Higher population density keep the pressure on forests, resulting in less forest-cover gain. In particular, a larger agricultural population will be associated with less forest area.

3. Roads (an indicator of economic development) have negative effects on forest cover. Larger road networks may not only cause forest losses directly through road construction, but also increase accessibility to forests, thus increasing forest losses.
4. Conservation policies have positive effects on forest-cover change. Plantations and natural regeneration occur due to the implementation of both conservation policies (NFCP and GTGP), which will increase forest cover.

Data Collection and Analysis

MODIS Products

Remote-sensing techniques constitute useful tools for monitoring land-cover change. Sensors such as the *Satellite Pour Observation De La Terre* (SPOT) and Landsat can provide a vantage point at medium spatial resolutions (~10 - 30 m) that are well suited for evaluating forest-cover change at regional scales (Jin and Sader 2005). However, relatively few cloud-free products are available from these systems due to their low temporal resolutions, which hinder forest monitoring at large scales and during long periods (NASA; Morisette et al. 2002). Coarse resolution sensors (e.g. 250-1000 m) may overcome these limitations by providing cost effective, frequent coverage of the earth's surface (Zhan et al. 2002).

As a key instrument onboard the Earth Observation System (EOS) Terra satellite, the first Moderate Resolution Imaging Spectroradiometer (MODIS) instrument was successfully launched in December 1999. The first Earth images from MODIS were obtained in February 2000. The second MODIS instrument was integrated on EOS Aqua satellite and successfully launched in May 2002 (Justice et al. 2002). The MODIS sensor includes 36 spectral bands ranging from the visible to the thermal infrared wavelengths (between 0.405 and 14.385 μm).

The first seven bands are designed primarily for remote sensing of the land surface with spatial resolutions of 250 m (bands 1 and 2), and 500 m (bands 3 to 7), which are centered at ca. 648 nm, 858 nm, 470 nm, 555 nm, 1,240 nm, 1,640 nm, and 2,130 nm, respectively (Justice et al. 2002). Besides the wide range of spectral coverage, the MODIS sensor also features various temporal resolutions; it can provide daily, 8-day, 16-day, monthly, quarterly, and yearly data to meet different research needs. MODIS provides more opportunities to address questions related to biogeochemical cycling, energy balance, land cover, and ecosystem change at regional and global scales (Justice et al. 1998; Jin and Sader 2005).

The MODIS Land Discipline Team (MODLAND) has produced a series of products relevant to earth system science and global change research (Wu et al. 2008; Ran et al. 2010), which include: (1) Radiation Budget Variables: Surface Reflectance, Land Surface Temperature (LST)/Emissivity, Snow and Ice Cover, Albedo/Bi-directional Reflection Distribution function (BRDF); (2) Ecosystem Variables: Vegetation Indices, Leaf Area Index (LAI)/Fractional Photosynthetically Active Radiation (FPAR), Vegetation Production: Daily Photosynthesis (PSN)/Annual Net Primary Production (NPP); and (3) Land Cover Characteristics: Fire and Thermal Anomalies and Burned Area, Land Cover, Vegetative Cover Conversion, and Vegetative Continuous fields. All these products are produced from MODIS raw data and are freely distributed to the public (LPDAAC; Masuoka et al. 1998). The MODIS Land Cover Type product in particular, integrates multiple classifications, which describe land-cover properties derived from observations spanning a year's input of MODIS data. The primary land cover classification scheme identifies 17 land cover classes defined by the International Geosphere Biosphere Programme (IGBP), which include 11 natural vegetation classes, 3 developed land classes, and three non-vegetated land classes (LPDAAC) .

I used the MODIS Land Cover Type product (MCD12Q1 Yearly L3 Global 500 m SIN Grid) for assessing forest-cover changes in the study area. This product contains five different land cover classification schemes: (1) the International Geosphere Biosphere Programme (IGBP) global classification scheme, (2) the University of Maryland (UMD) scheme, (3) the MODIS-derived LAI/fPAR scheme, (4) the MODIS-derived Net Primary Production (NPP) scheme, and (5) the Plant Functional Type (PFT) scheme (LPDAAC). Among these multiple classification schemes, only the IGBP scheme was applied in this study. This classification scheme includes 17 classes (i.e., water, evergreen needleleaf forest, evergreen broadleaf forest, deciduous needleleaf forest, deciduous broadleaf forest, mixed forest, closed shrublands, open shrublands, woody savannas, savannas, grasslands, permanent wetlands, croplands, urban and built-up, cropland/natural vegetation mosaic, snow and ice, and barren or sparsely vegetated). Different land-cover types are assigned different values from 0 to 16. This scheme was chosen for this study for the following two reasons: (1) It is specifically designed for the improvement of large-scale vegetation models needed for global and regional assessments (IGBP 2006), and (2) it has been proven to exhibit high accuracy in identifying land-cover types in China, especially after aggregating the original 17 classes into few combined land-cover classes (Wu et al. 2008; Ran et al. 2010).

For this study, eight consecutive years of IGBP products (from 2001 to 2008) were downloaded from the Land Processes Distributed Active Archive Center (LPDAAC) (https://lpdaac.usgs.gov/lpdaac/products/modis_products_table). The 17 different land-cover classes of the IGBP classification scheme were reclassified into two categories (forest and non-forest). For this, all forest types (i.e. evergreen needleleaf, evergreen broadleaf, deciduous needleleaf, deciduous broadleaf and mixed forests) were merged into the forest category, while

the other classes were merged into the non-forest category.

To evaluate the accuracy of the forest/non-forest reclassification of the IGBP scheme, I used 425 ground-truth plots collected in both study regions (Qinling and Sichuan Giant Panda Sanctuary) from 2004 to 2007. These plots consisted of a circular area with a 10-m radius, where the land-cover type was recorded. Among the 425 ground-truth plots, 141, 159, 40 and 85 points were collected in the four consecutive years from 2004 to 2007, respectively, with 175 and 250 points collected in the Qinling Mountains and the Sichuan Giant Panda Sanctuary, respectively. The user's accuracy for non-forest and forest areas was 84 and 87% respectively, and the overall accuracy was 87% (Table 1). Thus, the IGBF land-cover product merged into forest/non-forest cover can be used for detecting forest-cover change in the study regions.

MODIS Product Data Analysis

The digital version of the County Administration Maps of all counties in the two study regions were downloaded from the State Bureau of Surveying and Mapping (<http://www.sbsm.gov.cn/>), which were used to digitize township boundary files. Township boundary maps were produced by using ArcGIS 9.2 software (ESRI) to digitize the boundary of each township in County Administration Maps. Forest-cover information from each township was extracted from the MODIS Land-Cover product by using a corresponding digitized township-boundary file. The proportion of each township under forest cover in each of the eight years available (2001 through 2008) was calculated by dividing the number of forest pixels by the total number of pixels within each township's boundary. Considering the inter-annual variability observed in the MODIS Land-Cover Type product, a linear-regression analysis was employed on a per-township basis to detect the trend of forest-cover change between 2001 and 2008. If the regression line showed a significant ($p < 0.05$) trend (either positive or negative) of

percent forest-cover change with respect to time (i.e., year), then the values of forest-cover change (Δ forest) between 2001 and 2008 were calculated per township based on the empirical equation obtained by each regression analysis. On the contrary, if the regression line showed a non-significant trend, the difference of forest cover change between 2001 and 2008 was considered as 0. Examples of these trends in three different townships are shown in (Figure 3).

Evaluation of the Driving Forces of Forest Cover Change

Independent Variable Selection

Many previous studies have explained the dynamics of forest-cover change at different scales by combining remote-sensing data with spatially-explicit biophysical and socioeconomic information (Pahari and Murai 1999; Mertens et al. 2000; Gautam et al. 2004; Ali et al. 2005; Armenteras et al. 2006; Chowdhury 2006; Ferreira et al. 2007). Here, I used the results of some of these studies to select potential socioeconomic and biophysical variables that could have a direct influence on deforestation and/or forest transition. Many factors have been reported as underlying driving forces of forest-cover change. For instance, human population is often considered a determinant factor of forest-cover change. Allen and Barnes (1985) found a statistically-significant relationship between deforestation and human population growth (Allen and Barnes 1985). Moreover, population decline also results in abandonment of cropland, reductions in newly-reclaimed lands and livestock production, which ultimately contribute to reforestation (Douguedroit 1981). In addition, rural population, particularly in developing countries, depends on fuelwood for cooking, heating, and construction (Deweese 1989; Krutilla et al. 1995), thus the size of rural population is considered a cause of deforestation (Southgate et al. 1991). Agricultural population is used as a proxy of rural population. Household size is also considered an important factor, because smaller households have higher per-capita resource

consumption, which may impact forest-cover change and biodiversity (Liu et al. 2003). Although population dynamics are closely related to deforestation and reforestation, these demographic factors are not the only drivers of forest-cover change (Southgate et al. 1991). Technical developments in agriculture also contribute to forest-cover change. Increased agricultural yields result in conversion of marginal cropland to forest (Mather 1992). In addition, countries that rarely invest in agricultural development and that make no progress in agricultural productivity may have higher probability of deforestation (Mather 1992). Therefore, in this study, the unit production of cropland was selected as a proxy of agricultural technical development. As an indicator of economic development, the road network is another driving force which may have impacts on forest cover (Krutilla et al. 1995). Roads have many ecological impacts on the vegetation, wildlife, and humans close to them, particularly in increasing accessibility to remote areas and allowing various human activities (logging, hunting, and transportation) (Barnes 1990; Conway et al. 2000; Freitas et al. 2010).

Besides these socioeconomic factors, a large body of literature also shows that biophysical factors play important roles in forest-cover change (Barnes 1990; Bhattarai et al. 2009; Freitas et al. 2010). A study conducted in Africa found that the area of remaining forest is an important variable determining the rate of forest loss (Barnes 1990). Moreover, topography is also a determinant factor of land use and forest distribution. Studies in Brazil and Nepal found that topography (e.g. slope and elevation) can directly affect deforestation (Bhattarai et al. 2009; Freitas et al. 2010). Besides demographic, economic, and biophysical factors, government policies also play an important role in influencing forest-cover change, as has been demonstrated in several places within China (Zhao et al. 2000; Liu and Edmunds 2003; Liu et al. 2008). Finally, forest-cover change exhibits regional differences due to differences in the underlying causes.

Therefore, in this study regional differences between the Sanctuary and the Qinling Mountains were tested using the region as a dummy variable in statistical models of deforestation (Bawa and Dayanandan 1997).

Variable selection in this study was based not only on the approaches undertaken in other studies, but also on the availability of data for the entire study area. Further details on specific variables, their collection procedures and measurements considered in this analysis are provided below.

All demographic, economic, and conservation data were obtained from government documents and personal interviews, which were used as independent variables in a regression model. Both demographic and economic data were acquired from the statistical yearbooks of each county. Information on the implementation of the NFCP and GTGP was obtained from government sources (i.e., Forestry Bureau, Center of NFCP and Office of GTGP) in each county, and from published reports of the NFCP and GTGP implementation, when available. These reports contain information such as implementation date, total planned area, and total implemented area. Since the NFCP was implemented by both the forestry bureau of each county and some state-owned forestry bureaus (major timber producers before the logging ban), I visited the Departments of Forestry of Shaanxi and Sichuan Provinces to collect information on the NFCP implementation by these state-owned forestry bureaus. In addition, 22 related county and township government officials were interviewed face-to-face in order to obtain additional information and in order to evaluate the quality of the government reports through comparing the consistency of the information provided by government officials with that obtained from government reports. Government reports seem to be reflecting the real status of the implementation of both conservation policies.

Topographic data (i.e. elevation and slope) were obtained from a Digital Elevation Model (DEM) derived from the Shuttle Radar Topography Mission (SRTM) (Rabus et al. 2003). Since the township area is much larger than the pixel size of the SRTM data ($90\text{m} \times 90\text{m}$), the data on elevation and slope were averaged by township.

The following explanatory or independent variables were included in the final regression model:

POPDEN — population density of a township (individuals/km²); calculated mean of human population during 2001 to 2008 of each township, divided by area of the corresponding township.

PAPOP — percentage of agricultural population (%); calculated mean of agricultural population during 2001 to 2008 of each township, divided by mean of total population size during the same period of the corresponding township;

HHSIZE — average household size (number of individuals/household); calculated mean of population during 2001 to 2008 of each township, divided by mean of number of households during the same period of the corresponding township;

PCROPL — percentage of cropland area within the township boundary (%); calculated mean of area of cropland during 2001 to 2008; divided by total area of the corresponding township;

UPCROP — unit production of cropland [ton/mu (1 mu = 1/15 ha)]; calculated mean of annual production of cropland during 2001 to 2008 of each township, divided by mean of area of cropland of the corresponding township during the same period;

FOR2001 — forest cover in year 2001(%); percentage of forest pixels within the township boundary;

ELEVAT — average elevation of the township (m);

SLOPE — average slope of the township (degree);

RDDEN — road density of each township (km/km^2); total length of roads within township boundary, divided by the total area of the corresponding township;

GTGP — percentage of the GTGP area within a township boundary (%); calculated total implemented area of the GTGP, divided by total area of the corresponding township;

NFCP — NFCP implementation status; a dummy variable with 0 representing non-NFCP implementation and 1 NFCP implementation;

REGION — regional difference; a dummy variable with 1 representing townships in the Qinling Mountain region, and 0 representing townships in the Giant Panda Sanctuary.

Collinearity Diagnostics

Multicollinearity among the selected predictor variables is a problem often encountered when applying multiple regression models (Lipovetsky and Conklin 2001; Ott and Longnecker 2001). In order to reduce the impacts of multicollinearity on individual variables of the regression model, a collinearity diagnostic was conducted. The Variance Inflation Factor (VIF) and tolerance are both widely-used measures of the degree of multi-collinearity of the i th independent variable with the other independent variables in a regression model (O'Brien 2007). The VIF and tolerance are defined as (Ott and Longnecker 2001; O'Brien 2007):

$$\text{Tolerance} = \frac{1 - R^2_{x_j \cdot x_1 \cdots x_{j-1} \cdot x_{j+1} \cdots x_k}}{1}$$

$$VIF = \frac{1}{\left(1 - R_{x_j \cdot x_1 \dots x_{j-1} \cdot x_{j+1} \dots x_k}^2\right)}$$

Where $R_{x_j \cdot x_1 \dots x_{j-1} \cdot x_{j+1} \dots x_k}^2$ is the R^2 value obtained by letting x_j be the

dependent variable in a multiple regression, with all other x_k independent variables. In addition,

multiple correlation is also a straightforward method of assessing the amount of

multi-collinearity in a set of independent variables (Ott and Longnecker 2001). Here, both

methods were combined to make a final selection of variables to be used. A common rule of

thumb is that if VIF is higher than 5, then multi-collinearity is of concern (O'Brien 2007).

Moreover, based on my knowledge of the study regions and a correlation matrix of all variables

(Table 3), the first three variables (i.e. SLOPE, PCROPL and ELEVATION) exhibiting higher

VIF values were excluded (Table 2). Population density is a direct indicator of human pressure

on land-cover change, which can reflect the level of human demand in my study regions.

Therefore, it is reasonable to substitute population density for percentage of cropland in the

model. Second, slope and elevation had higher correlation with region and both variables are

significantly different between the two regions ($p < 0.01$). However, within each region, both

topographic characteristics are relatively consistent. So region can be the proxy of topographic

difference in the two study regions. Except these three excluded variables, the remaining

variables were used in the regression model.

Regression Analysis

An Ordinary Least Squares (OLS) Regression model was employed to estimate the effects of

socioeconomic factors, biophysical factors, and conservation implementation status (GTGP and NFCP) on the forest-cover change. In the model, the dependent variable was forest-cover change at the township level between 2001 and 2008.

OLS regression is defined as

$$y = \beta X + \varepsilon$$

Where y is the dependent variable, β is the vector of coefficients, X is the set of dependent variables, and ε is the vector of random error terms.

Since statistical yearbooks of 11 townships (5 townships in Lushan County and 6 townships in Dujiangyan County) in Sichuan Panda Sanctuary were not available and the 10 townships in Zhouzhi County of Shaanxi Province are located in flat areas and with little forest cover present within their boundaries, a total of 108 townships (83.7 % of all townships in the two study regions) in 13 counties were selected for the final OLS model. After considering the results of multi-collinearity diagnostics, the specification of the regression model was as follows:

$$Y = \beta_0 + \beta_1 X_1 + \beta_2 X_2 + \cdots + \beta_9 X_9 + \varepsilon$$

Where, Y is the forest-cover change between 2001 and 2008 for each township, and X_k stands for the explanatory variables POPDEN, PAPOP, HHSIZE, UPCROP, RDDEN, GTGP, NFCP, REGION and FOR2001, respectively; and ε is the term representing the vector of random errors.

RESULTS

Across the entire study areas, after eight years of GTGP and NFCP implementation, the non-forest areas decreased in many townships and the forest areas expanded (Figure 4). Visually, many forest patches connected with each other and many non-forest patches disappeared. However, in both study regions, besides areas with forest gain (light green), forest loss can still be observed, particularly in the lower part of Qinling Mountains region.

Results from the regression analysis of forest-cover change vs. time at the township level showed that 43% of the townships (i.e., 46 of 108) had a significant increase while 7% (8 of 108) had a significant decrease in forest-cover change from 2001 to 2008. The remaining townships (54 of 108) did not show significant changes in forest cover (Figure 5 and Figure 6). Most of the townships in the Sichuan Giant Panda Sanctuary exhibited forest-cover gains, while in the Qinling Mountains, most of the townships showed no change in forest cover. In addition, most of the townships that exhibited significant forest-cover losses were located in the Qinling Mountains (Figure 5).

All townships in the study area are partly or entirely located in the mountainous region and covered by one or both conservation policies. There are 13 townships covered by one conservation policy, including 12 townships that are only covered by the GTGP and 1 township only covered by the NFCP. All other townships are covered by both conservation policies. Among the 13 townships with single conservation policy implementation, six showed forest-cover losses from 2001 to 2008, and two showed stable forest cover. Based on this small sample size, it is not possible to evaluate if forest-cover change is affected when both conservation policies are implemented simultaneously as opposed to only one. However, it is

hypothesized that in some areas the implementation of a single policy may produce less forest-cover gain due to a smaller implementation area compared to the areas involved when both policies are implemented simultaneously.

Table 4 shows the results of the OLS model. Among the five socioeconomic variables, population density (POPDEN), percentage of agricultural population (PAPOP), and road density (RDDEN) had a significant negative effect on forest-cover change. The initial forest cover in 2001 (FOR 2001) also showed a significant negative effect on forest-cover change. In contrast, both conservation policy variables had significant positive effects on forest-cover change.

DISCUSSION

Results of the linear regression model show that forest-cover change is best explained by multiple factors acting synergistically rather than by single-factor causation, which is in accord with many other studies about causes of forest-cover change (Burgess and Sharpe 1981; Southgate et al. 1991; Bawa and Dayanandan 1997; Geist and Lambin 2001). The township-level forest-cover change map (Figure 5) shows that although both regions started implementing conservation policies at about the same time, they had quite different recovery patterns and processes. Most of the townships in the Sichuan Giant Panda Sanctuary experienced forest-cover gains, while most of the townships in the Qinling Mountains showed unchanged forest cover. Historically, most changes in forest cover result from human activities. The demographic variables are widely accepted as important drivers of forest-cover change worldwide (Houghton 1991; Meyer and Turner 1992; Jorgenson and Burns 2007; Carr 2008). In this study, various demographic pressures may cause significant differences in forest-cover change between the two study regions. Different population densities may result in different forest-recovery status in the two regions. Since township population density in the Qinling Mountains is significantly higher than in the townships of Sichuan Giant Panda Sanctuary ($p < 0.01$), forests in the Qinling Mountains region may face higher human pressures than those in the Sichuan Giant Panda Sanctuary.

Moreover, this study also showed that demographic characteristics are important determinants of forest-cover change, having significantly negative effects. The results support the negative relationship between forest cover and population pressure reported by previous studies (Allen and Barnes 1985; Carr et al. 2005; Jha and Bawa 2006). The change in population

alters the demand for land and forests, which are expected to supply food, fuel, and environmental services for local people (Mikesell 1960; Allen and Barnes 1985; Williams 1989). Before conservation policy was implemented, larger populations meant more demands on natural resources (e.g. timber, fuelwood, and forestry products) and more land conversion from forest to non-forest. After the year 2000, both the NFCP and GTGP were implemented in these regions, and logging was strictly banned. Although large-scale commercial timber harvest has ceased and illegal logging has been controlled (Zhang 2006), higher population pressures may reduce the forest-cover gains. In order to meet the demands of local people, fuelwood consumption and timber used in new housing construction still has a significant impact on forest restoration (Liang 2008).

Moreover, a higher percentage of agricultural population usually causes greater dependency on fuelwood for cooking and heating (Krutilla et al. 1995). In addition, other activities, such as cultivation and grazing conducted by local agricultural population, may also offset part of the forest gains brought about by conservation policy implementation.

Besides demographic variables, roads also had a significantly negative impact on forest-cover change in my study areas. Many previous studies indicated that roads are considered an important factor behind deforestation and forest fragmentation (Young 1994; Pfaff 1999; Nagendra et al. 2003; Soares et al. 2004; Fearnside 2007; Fearnside 2008). Roads not only directly impact forest resources through road construction (Laurance et al. 2009), but also have a key role in facilitating the exploitation of forest regions (Ali et al. 2005; Pfaff et al. 2007; Laurance et al. 2009). Road networks, which provide market access to remote areas, may induce local people to exploit more natural resources. Moreover, road construction for timber harvesting sometimes provides access for agricultural populations in search of new land to clear and farm

(Allen and Barnes 1985; Southgate et al. 1991; Nepstad et al. 2001; Soares et al. 2004; Fearnside 2007). The road networks within the study area may have mixed effects on forest-cover change. On the one hand, widening old roads and construction of new roads may directly cause forest-cover losses. In both study regions, a new round of road-network extension projects named “roads to every village” has been conducted since 2005 (SCGOV 2006). Many counties in both study regions have achieved the goal of connecting every village with paved roads (Personal observation). On the other hand, with growing road networks and improved road conditions, many townships in both study regions have developed or are planning to develop tourism activities (Li and Han 2001; Fang 2002; Li 2004; He et al. 2008; Luo and Zheng 2008). For example, the number of tourists in Wolong Nature Reserve, one of the most important giant panda nature reserves in China and located in the Sichuan Giant Panda Sanctuary, increased from 130,000 in 2000 to 206,100 in 2005 (He et al. 2008). The booming tourism causes more wood consumption in both study regions, since more fuelwood is required to meet demands of tourists (e.g. cooking, heating, making smoked pork) (Gaughan et al. 2008) and many conventional tourism facilities (e.g. restaurants, hotels) have been constructed, which usually consume fuelwood (Field Observation).

The initial forest cover often determined the level of implementation of conservation policies. In order to preserve species and their habitat or prevent flooding and soil erosion, local governments are under pressure to protect the remaining forests and implement conservation policies (Grainger 1995). Forest plantation is a fast and direct way to help forest recovery and reach the transition point (from forest-cover loss to forest-cover gain). In townships with a larger area of clear-/selectively-cut areas, barren lands, or sloping croplands it may be a priority to apply reforestation and afforestation methods such as tree planting and aerial seeding. On the

contrary, in townships with larger areas of remaining forest, forest protection and surveillance will be applied first, and forest restoration will rely mainly on natural regeneration. Since reforestation and afforestation usually produce faster effects than natural regeneration during a short period, the townships with lower initial forest cover may have relatively greater forest-cover gains as compared with other townships with larger initial forest cover.

Contrary to the effects of the previously described variables, both conservation policies had positive effects on forest-cover change in the study area. In mountain regions, cropland is a major non-forest land cover type. The GTGP helps local farmer households to convert steep-slope cropland into ecological or economic forests. This program appears to contribute to forest-cover gains in most townships with the GTGP implementation. Usually, fast-growing local tree species are selected for tree planting under the GTGP. These planted tree species may not only benefit the environment (e.g. reduce soil erosion and increase tree cover) but also increase the income of enrolled local farmer households within a relatively short period (Zhang et al. 2003). In addition, substantial labor supplies have been released from agriculture and attracted to more urbanized regions through rural-urban labor migration. Generally, temporary and permanent rural-urban migration is promoted by better job opportunities and living conditions in urbanized areas and cities (Liang 2001; Li and Zahniser 2002), which not only enhance the conversion of abandoned marginal croplands to forest, but also reduce the pressure of local populations on natural resources (Chen 2010).

Compared with the single method (i.e. tree planting) of the GTGP, The NFCP consists of four different forest restoration/conservation methods (i.e. forest conservation and management, mountain closure, tree planting and aerial seedings). Although I cannot distinguish which method contributes more to forest-cover gains, generally, tree planting and aerial seedling may have

direct and faster effects on forest restoration during short periods (i.e. 8 years) than forest conservation and management (e.g. forest surveillance and anti-illegal logging patrols) and mountain closure. However, over the long run, all methods will show their effects on forest-cover gain in NFCP implemented areas.

While variables such as GDP per capita or net income per capita have also been identified as important drivers of forest-cover change in previous studies (Koop and Tole 1999; Mather et al. 1999; Ehrhardt-Martinez et al. 2002; Bhattarai and Hammig 2004; Culas 2007), they were not included in this study due mainly to data limitations. However, future studies should consider them in order to evaluate their effects on forest-cover change in the study regions.

Although conservation policies have shown positive effects on forest cover not only in my study regions, but also in all of China's forest and society (Liu et al. 2008), we should not ignore their potential global environmental implications. Today, China has become a world-leading timber importer and wood product exporter (EIA 2007). Implementation of forest conservation policies in China have raised global concerns that as a result of these policies, China's timber import is exerting enormous pressures on the forests of other regions such as South East Asia (e.g. Burma and Indonesia), Madagascar, and eastern Russia, often in the form of illegal logging (Laurance 2008; CIFOR 2010). Future research, therefore, needs to pay close attention to the global effects of China's domestic forest conservation/restoration policies.

CONCLUSIONS AND RECOMMENDATIONS

Based on the information extracted from the MODIS Land Cover Type product (from 2001 to 2008), this study shows that forest regeneration has happened in most of the study regions. Based on the regression model, my study also suggests that demographic factors have significant negative effects on forest-cover change. Specifically, population density and percentage of agricultural population were strongly and negatively related to forest-cover change. As a proxy of development, road density was also significantly negatively related to forest cover change. Extension of road networks may not only directly result in forest-cover losses but also increase wood consumption caused by tourism development and other types of forest exploitation. Although in some areas, road networks increase access to other types of energy (e.g. coal and electricity), in my study area, the use of coal and electricity as an alternative to fuelwood is still low (Wang et al. 2010).

In order to mitigate the negative effects caused by the above driving forces and enhance the positive effects of conservation policies on forest-cover change, local governments can implement two additional actions. On the one hand, they should help local households switch their energy sources from fuelwood to others, such as electricity and methane. Generally, methane has the advantages of being cheap, easy to generate, and multifunctional, as it can be generated by fermentation of human and livestock waste, or of corn and/or wheat stalks, thus providing energy for cooking and heating. For the households that cannot switch from fuelwood to other energy sources, the government and non-governmental organizations (e.g. World Wide Fund for Nature) may help them change their stoves to fuelwood-saving types. Fortunately, both study regions have started to use these strategies to reduce the negative effects on forest-cover

change (WWF 2004; SFA 2009d). On the other hand, besides energy substitution strategies, rural-urban labor migration may also reduce human impacts on forests. For this, local governments should provide training for local people in order to increase their skills to obtain job opportunities in urban areas.

After eight years of implementation, conservation policies seem to have achieved their expected effects in terms of restoring forest cover and conserving natural forests (Liu et al. 2008). Most townships in my study areas exhibited either forest regeneration or have effectively protected their remaining forests. The GTGP was recently renewed for another eight years when most of its initial contracts started to expire in 2008. Continuous conservation efforts through the GTGP and the NFCP are important for preventing deforestation and forest degradation in China.

The results of this study have two limitations. First, since all GTGP plots are smaller than one MODIS pixel, some forest-change information may be lost due to the relatively coarse spatial resolution of data generated by this system. Second, the OLS regression model used is based on the assumption that all variables included in the model are completely independent. While interactions among variables are possible, this effect was reduced by excluding some variables in order to reduce multi-collinearity. However, important effects of some excluded variables, as well as variables not included, could have been missed. Therefore, future analyses of forest cover change in the study regions should allow for a longer time period after the implementation of conservation policies in order to allow for a more noticeable change in forest cover (e.g., extending to areas larger than a MODIS pixel). They should also include additional variables that constitute potential drivers of forest-cover change in order to obtain a better picture of the mechanisms (in addition to conservation policies) affecting the dynamics of forest cover in the regions.

Forest transition in China is not a unique case in Asia. India and Vietnam have also undergone forest transition during the last decade (Foster and Rosenzweig 2003; Meyfroidt and Lambin 2009). In addition, other developing countries around the world have slowed down deforestation and may step into a forest transition within the near future (Henson 2005; Wannitikul 2005). Therefore, it is necessary to understand the underlying driving forces of these observed patterns and their ecological effects, which may contribute to understanding a possible emerging trend that would have important implications for future forest resources worldwide.

Table 1. Error Matrix of Classification of Different Land Cover Types

		Ground Truth Points		
MODIS product		Non-forest	Forest	Row Total
	Non-forest	16	3	19
	Forest	53	353	406
	Column Total	69	356	425
User's Accuracy				
		Non-forest = 16/19 = 84%		
		Forest = 353/406 = 87%		
Overall Accuracy = (16+353)/425 = 87%				

Table 2. Values of Variance Inflation Factor (VIF) of All Pre-Selected Variables

Variable	VIF
SLOPE	9.82
PCROPL	7.01
ELEVATION	6.42
POPDEN	5.27
FOR 2001	3.69
REGION	2.85
PAPOP	2.42
UPCROP	2.13
RDDEN	2.09
HHSIZE	1.83
GTGP	1.81
NFCP	1.43
Mean VIF	3.90

Table 3. Correlation Matrix of Pre-Selected Variables

	POPDEN	PAPOP	HHSIZE	PCROPL	UPCROP	ELEVAT	SLOPE	RDDEN	GTGP	NFCP	REGION	FOR2001
POPDEN	1.0000											
PAPOP	-0.3840	1.0000										
HHSIZE	-0.3078	0.5123	1.0000									
PCROPL	0.8399	-0.1332	-0.1886	1.0000								
UPCROP	0.3339	-0.0598	-0.2081	0.3340	1.0000							
ELEVAT	-0.4763	0.0319	0.3296	-0.5517	-0.5534	1.0000						
SLOPE	-0.7447	0.0608	0.3107	-0.8513	-0.3858	0.7392	1.0000					
RDDEN	0.4405	-0.2331	-0.1831	0.4885	0.2654	-0.5412	-0.6167	1.0000				
PGTGP	0.2304	-0.0338	-0.2506	0.2967	0.1687	-0.5701	-0.4839	0.4087	1.0000			
NFCP	-0.3846	-0.0349	0.2018	-0.3645	-0.0725	0.3392	0.4588	-0.1505	-0.2235	1.0000		
REGION	0.3413	0.1716	-0.1944	0.4328	0.1176	-0.6807	-0.6217	0.2929	0.4453	-0.4028	1.0000	
FOR2001	-0.5750	0.0474	0.0802	-0.6675	0.1290	0.1003	0.5957	-0.3293	-0.2544	0.2649	-0.2525	1.0000

Table 4. Results of the OLS regression model at the township level

Variable Category	Variables	Coefficients	Std. Error	P-value
Socioeconomic Variables	POPDEN	-0.011	0.006	0.069*
	PAPOP	-16.650	8.120	0.043**
	HHSIZE	3.159	2.163	0.147
	UPCROP	4.957	7.614	0.517
	RDDEN	-0.014	0.006	0.029**
Biophysical Attributes	FOR 2001	-0.099	0.043	0.022**
	REGION	-0.817	2.167	0.707
Conservation Policies	GTGP	45.081	25.601	0.081*
	NFCP	5.180	3.021	0.090*
Adjusted R² = 0.129				

* 0.05<P < 0.1; **P < 0.05

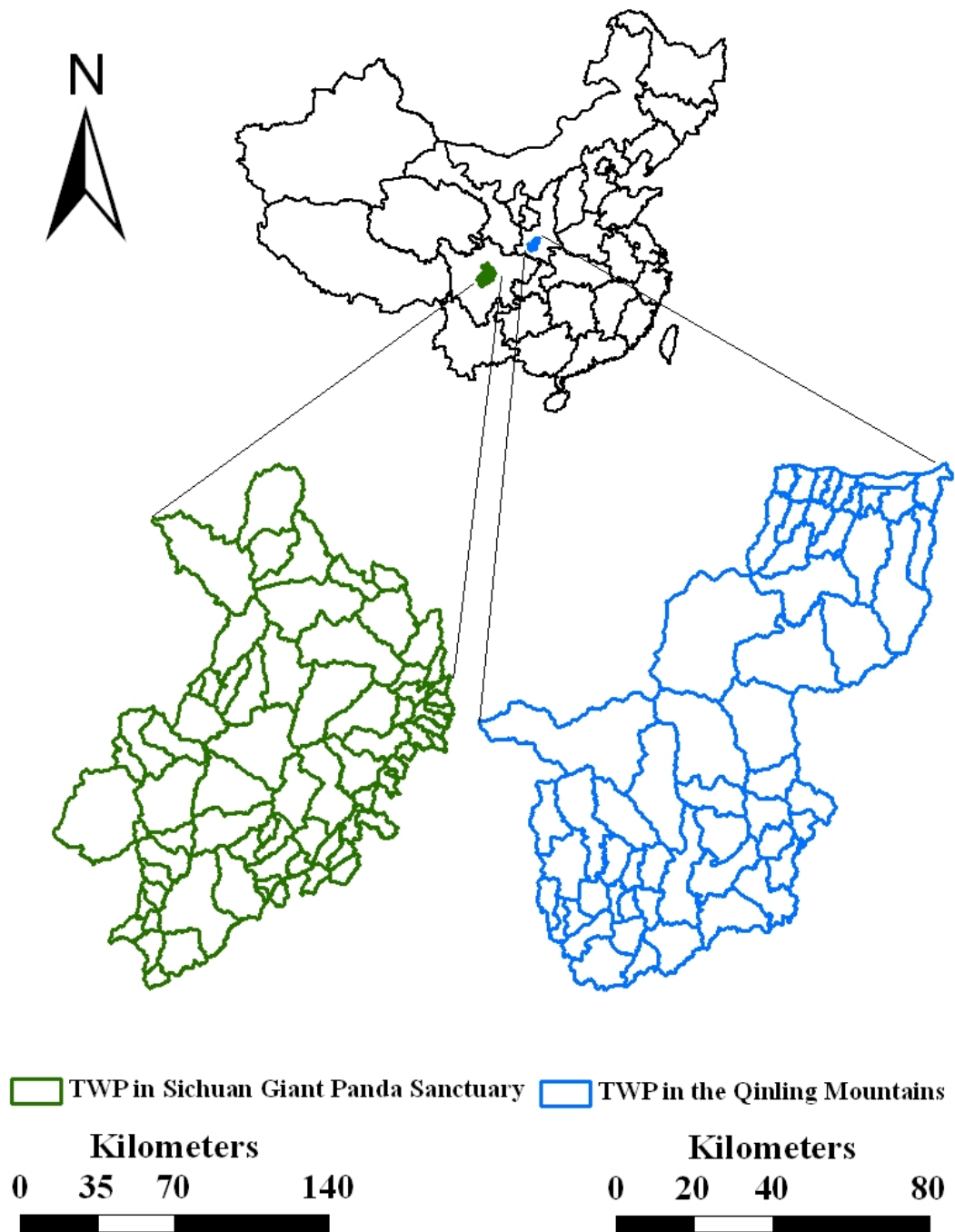


Figure 1. Two study regions and their locations in China (Left: Sichuan Giant Panda Sanctuary; Right: the Qinling Mountains). For interpretation of the references to color in this and all other figures, the reader is referred to the electronic version of this thesis

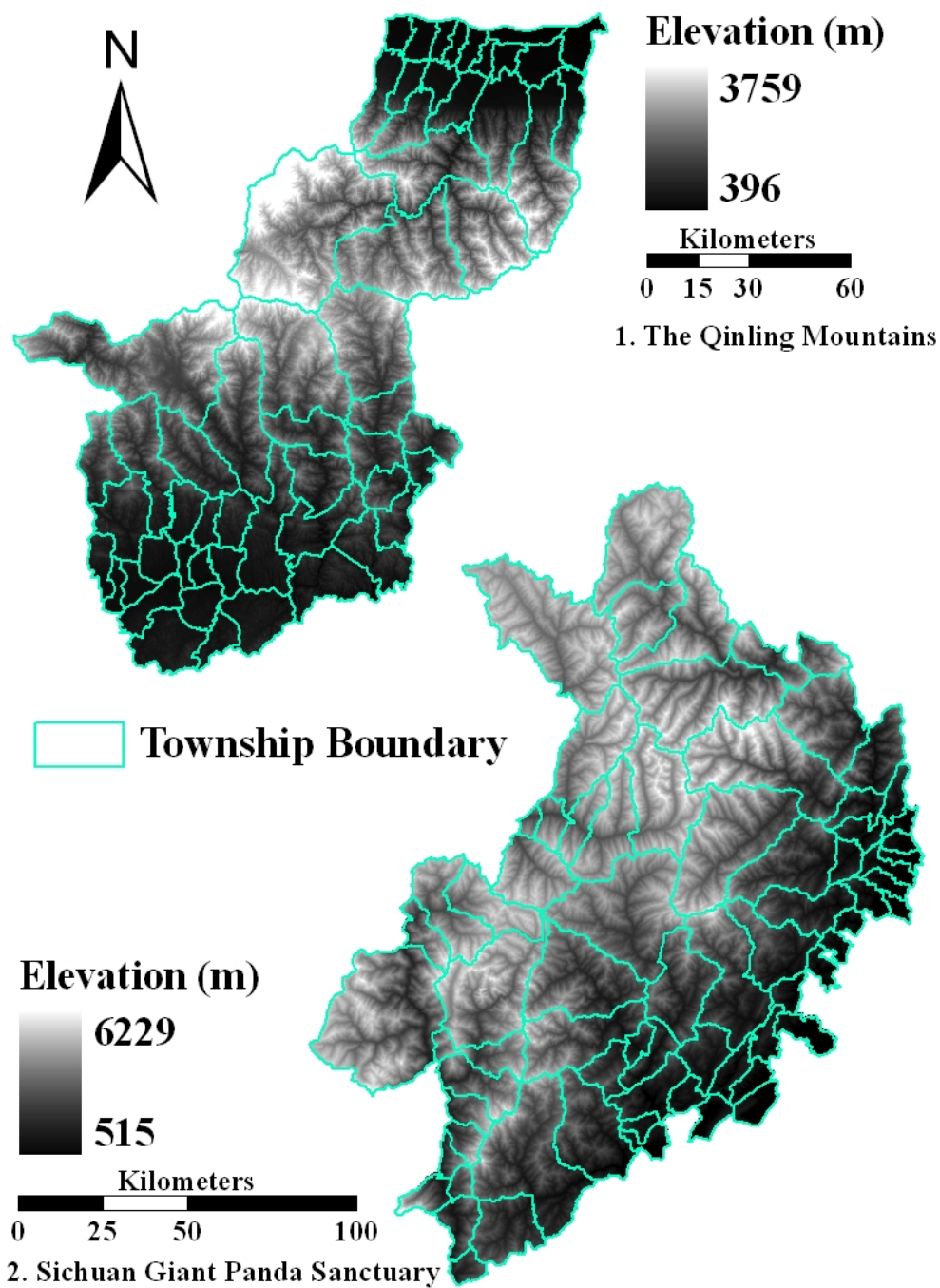


Figure 2. The elevation range of the two study regions (1. The Qingling Mountains; 2.Sichuan Giant Panda Sanctuary)

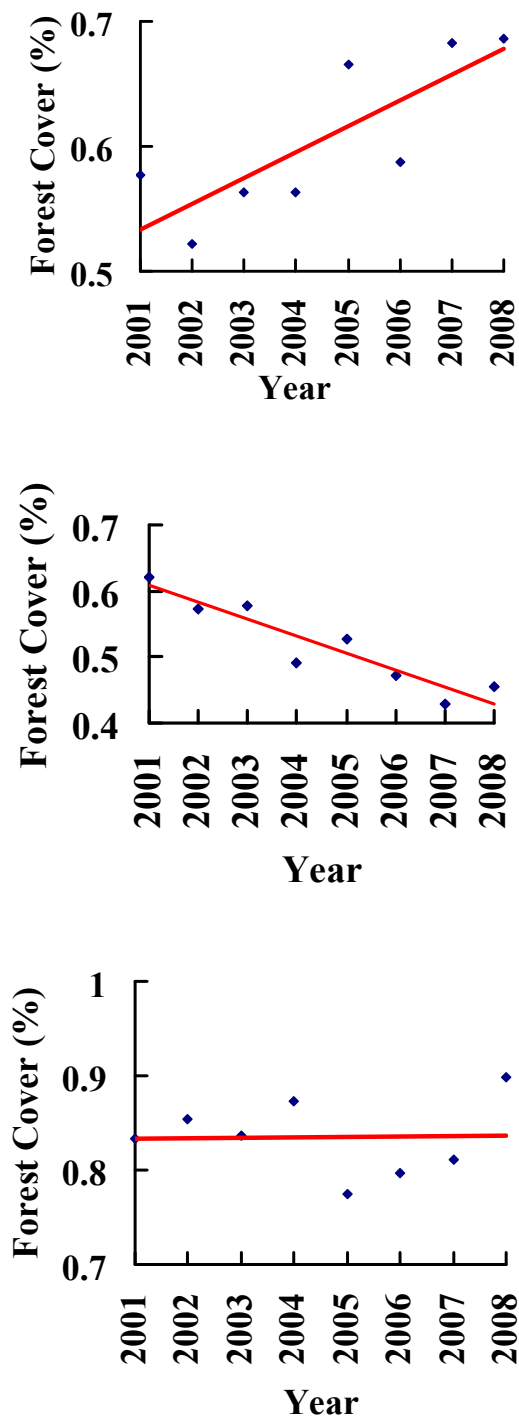


Figure 3. Examples of three types of forest-cover change [Top: Daheba Township had a significant increase (+ Δ forest); middle: Shiguan Township had a significant decrease (- Δ forest); bottom: Wushan Township had insignificant change (Δ forest = 0)]

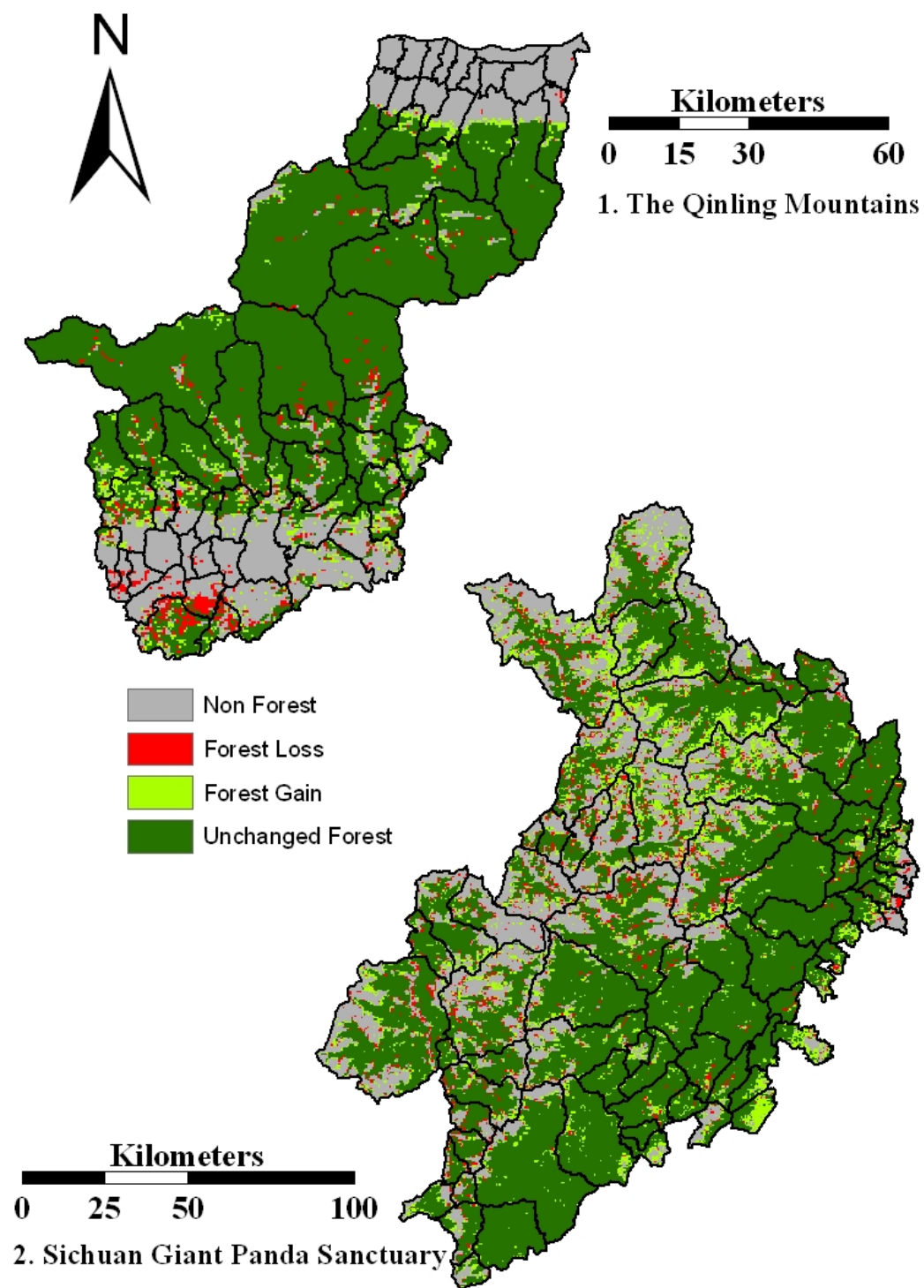


Figure 4. Forest cover change at pixel level from 2001 to 2008 (1. The Qinling Mountains; 2. The Sichuan Giant Panda Sanctuary)

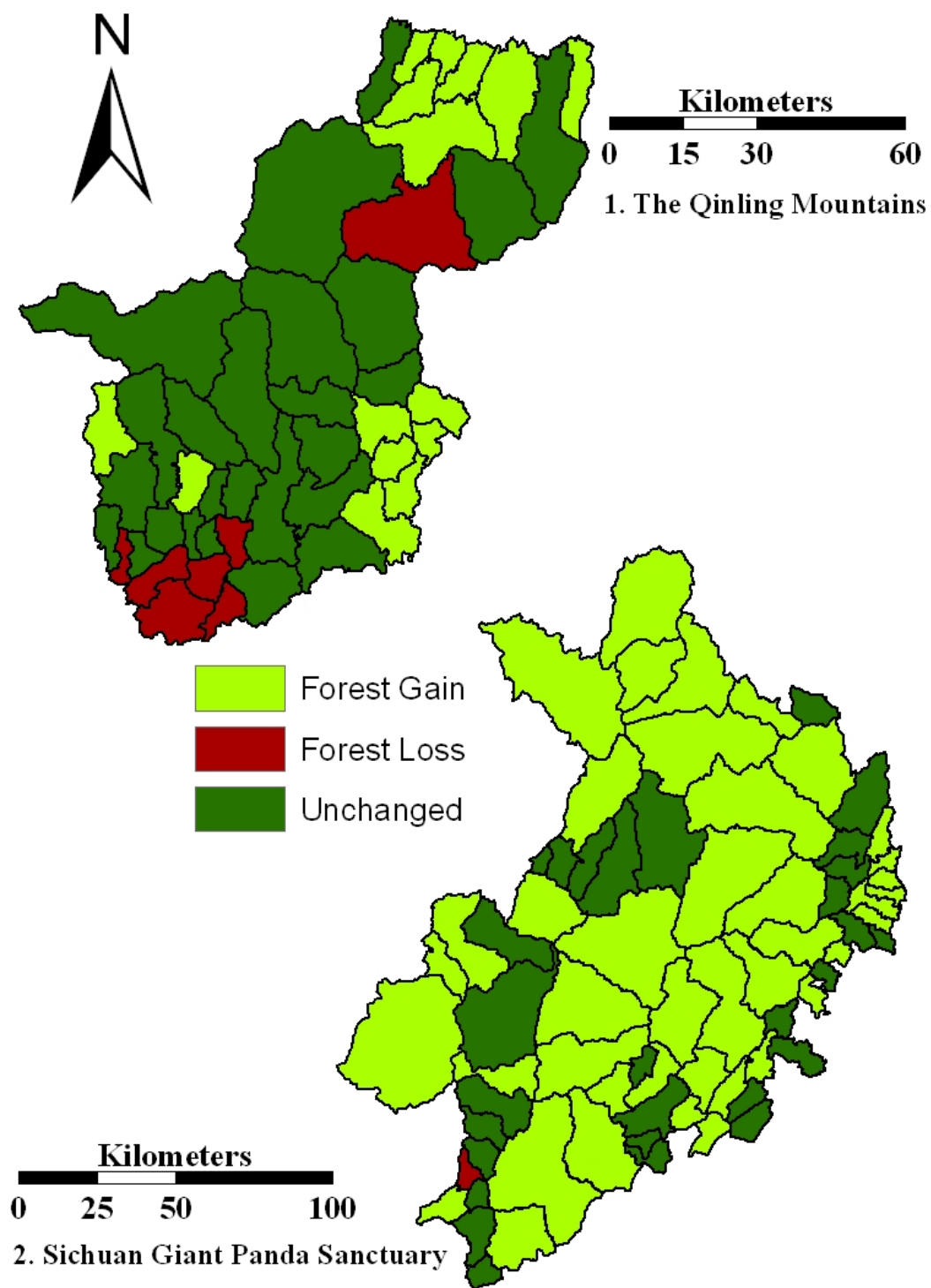


Figure 5. Forest-cover change at township level from 2001 to 2008 (1. The Qinling Mountains; 2. Sichuan Giant Panda Sanctuary)

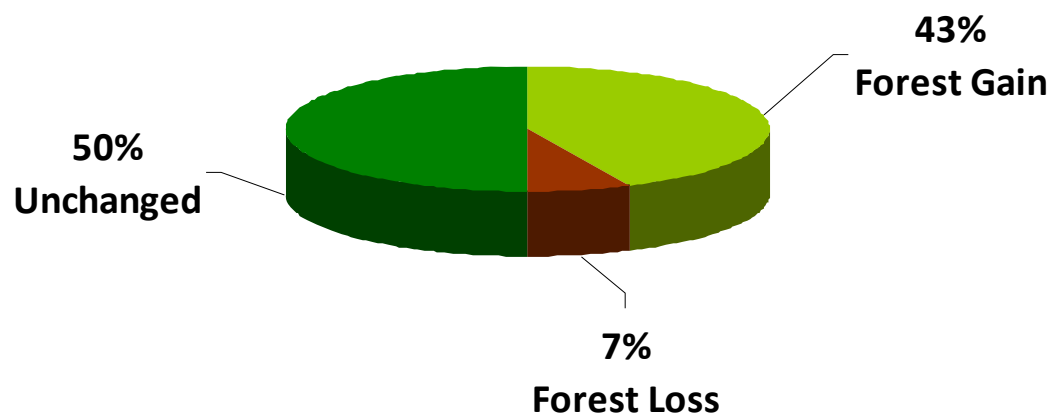


Figure 6. Percentage of townships with different forest cover change status

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