Biodivers Conserv (2009) 18:3421–3440
DOI 10.1007/s10531-009-9651-1

ORIGINAL PAPER

Clearance and fragmentation of tropical rain forest in Xishuangbanna, SW, China

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Received: 6 May 2008 / Accepted: 13 May 2009 / Published online: 24 May 2009

Abstract Xishuangbanna, situated in the northern margin of the tropical zone in Southeast Asia, maintains large areas of tropical rain forest and contains rich biodiversity. However, tropical rain forests are being rapidly destroyed in this region. This paper analyzed spatial and temporal changes of forest cover and the patterns of forests fragmentation in Xishuangbanna by comparing classified satellite images from 1976, 1988 and 2003 using GIS analyses. The patterns of fragmentation and the effects of edge width were examined using selected landscape indices. The results show that forest cover declined from 69% in 1976 to less than 50% in 2003, the number of forests fragments increased from 6,096 to 8,324, and the mean patch size declined from 217 to 115 ha. It was found that fragment size distribution was strongly skewed towards small values, and fragment size and internal habitat differ strongly among forest types: less fragmented in subtropical evergreen broadleaf forest, but severe in forests that are suitable for agriculture (such as tropical seasonal rain forest and mountain rain forest). Due to fragmentation, the edge width was smaller in 2003 than that in 1976 when the total area of edge habitat exceeded core habitat in different forest types. The core area of tropical seasonal rain forest was smallest among main forest types at any edge width. Fragmentation was severe within 12.5-km buffers around roads. The current forest cover within reserves in Xishuangbanna was comparatively large and less fragmented. However, the tropical rain forest has been degraded inside reserves. For conservation purposes, the approaches to establish forest fragments networks by corridors and stepping stone fragments are proposed. The conservation efforts should be directed first toward the conservation of remaining tropical rain forests.

Keywords Forest fragmentation · Edge effect · Core habitat area · Biodiversity · Nature reserves · Road buffer · Xishuangbanna

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Introduction

Disturbed by human activities, forest loss and habitat fragmentation have received worldwide attention (Pimm 1998; Laurance et al. 1997, 1998; Parthasarthy 1999). Habitat loss and fragmentation are among the principal causes of biodiversity loss and the collapse of primary productivity in the tropical rainforests (Ranta et al. 1998; Laurance et al. 1997, 1998; Debinski and Holt 2000; DeFries et al. 2005). Forest fragmentation includes both a reduction in interior habitat and an increase in edge length, edge habitat area and the degree of isolation of forest patches (Laurance et al. 2002; Cayuela et al. 2006). Forest fragmentation can severely modify habitat's physical or biota conditions that many species live on affect the distribution pattern of species, and even induce some species loss (Ma et al. 1998; Linera et al. 1998; Laurance et al. 1998; Cox et al. 2003).

Understanding the patterns and processes of habitat changes is essential for studying the relationship between forest habitat fragmentation, human impact, reserve networks, and biodiversity conservation (Revilla et al. 2001; Ferraz et al. 2005). Forest habitat sizes, extent of edge, and the past disturbance of remnant can strongly influence species responses to fragmentation (Debinski and Holt 2000; Euskirchen et al. 2001; Harper et al. 2005). Research on two comparable areas of evergreen montane forest reported that the areas with relatively large patches, some connectivity and few human activities could support populations of large mammals and frugivorous birds extirpated in other areas with small, isolated patches and more human interference (Pattanavibool and Dearden 2002). Researchers also found that species' abundances in fragments differed from those in intact forest, with some declining and others becoming hyperabundant (Laurance et al. 2002). Small (1–10 ha) and isolated fragments can lose species initially at a remarkably high rate (Laurance et al. 2002). Increased amount of edge habitat and edge influence are the most important consequence of fragmentation. The forest edge influences can lead to the degradation of forests fragments (Laurance et al. 2002). The edge effects of forest fragments are the most important proximate cause of elevated tree mortality, damage, and turnover by alterations in forest microclimate and greater wind turbulence near edges (Laurance et al. 1998). Evaluating edge effects within remnant forests and delineating area of edge influences of landscape are particularly important for resource assessments, biodiversity studies, landscape design, and wildlife habitat management (Zheng and Chen 2000). The more severe the pressure of human encroachment, the more fragmented the existing habitat becomes. Road construction as concentrated human activity promotes landscape modification (McGarigal et al. 2001). The ecological effects of roads can resonate to substantial distances from the road, creating habitat fragmentation and facilitating fragmentation through support of human exploitative activities (Trombulak and Frissell 2000). The increase in road density accounted for most of changes in landscape configuration associated with mean patch size, edge density, and core area metrics (McGarigal et al. 2001). Building protection area alleviated forest fragments and promoted biodiversity conservation (DeFries et al. 2005). Larger forest patches can provide effective protective habitat for species survival and is less susceptible to edge effects (Debinski and Holt 2000). Forest reserve designs frequently take into account fragmentation patterns to preserve larger and less isolated forest fragments (Ranta et al. 1998; Pattanavibool and Dearden 2002). Since small reserves might represent high quality remnants in tropical rain forest, they are also emphasized as protection targets for future expansion of reserve networks (Piessens et al. 2005). More studies have reported that the ecological consequences of fragmentation may differ depending on the patterns or spatial configuration imposed on a landscape and how it varies both temporally and spatially (Cayuela et al. 2006; Guirado et al. 2006).

Understanding the patterns of forest landscape change and the processes is essential for managing and conserving forest fragments and diversity conservation (Ranta et al. 1998; Laurance et al. 2002; Pattanavibool and Dearden 2002; Guirado et al. 2006).

In this study we analyze spatially explicit information on forest cover change and forest fragmentation state throughout Xishuangbanna, southwest, China, over the period 1976–2003. Xishuangbanna was one of the richest biodiversity spot in China (Cao and Zhang 1997). It represents only 0.2% of the area of China, but it contains approximately 5,000 species of higher plants (16% of the nation's total), 102 species of mammals (21.7%), 427 species of birds (36.2%), 98 species of amphibians and reptiles (14.6%), and 100 species of freshwater fish (2.6%) (Zhang and Cao 1995). With human population growth, traditional slash-and-burn agricultural activities and rubber plantation expansion, deforestation was dramatic in the past decades. Human population, increased from 220,000 in 1953 to 990,000 in 2000 in Xishuangbanna, suggested that the increase of population for exploiting the nature resource may be one of important factors affecting landscape change. On the other hand, rubber plantations development was a major threat to local primary forest. Large area of tropical rain forest and shifting cultivation lands at lower altitudes has been converted to rubber plantations over last 50 years, thereby inducing the clearance of forest distribution at high altitudes or steep slopes for new arable land demand (Li et al. 2007).

Nature reserves were built in Xishuangbanna as the loss of forest and biodiversity received local government's attention. At the same time, many forestry laws and policies related to forest conservation were formulated, such as Natural Forest Conservation Program in 1998 (Long et al. 1999), and Sloping Land Conversion Program in 1999 (Zhang et al. 2000). However, these policies had mixed impact within Xishuangbanna and natural forest cover was still declining. The total forests cannot reflect the status of habitats regarding biodiversity since rubber plantation is included in forests by definition. Although about 12% of the total area of Xishuangbanna are built for nature reserves (Guo et al. 2002), each sub-reserve had become an isolated island because most areas outside the protected area became farmlands and plantations (Fig. 1). Forest loss and fragments influence the species dynamics. For example, the incidents of human-elephant conflicts in Xishuangbanna often appeared in public media (http://news.qq.com/a/20090112/000106.htm). The main threats to the survival of Asian elephants are habitat alteration and availability of food caused by increased human interference (Zhang and Wang 2003; Feng and Zhang 2005). It is important to identify and understand Asian elephants' habitat structure, continuity of habitat, availability of food, and the movement patterns of herds to ensure the continued existence of Asian elephants in China (Zhang and Wang 2003). A research on species diversity change with tropical rain forest fragmentation also reported that the most dominant species (i.e., Barringtonia macrostachya) in primary nature rain forest in Xishuangbanna disappeared with forest fragmentation and the plant species diversity is generally lower in the fragmented forests than in the primary forest (Zhu et al. 2004). With natural primary forests cover decrease, local ecological environment was also degraded, such as fog formation and duration reduction (Huang et al. 2000; Liu et al. 2004, 2007).

Although some of the ecological consequences of forest fragmentation have been investigated in Xishaungbannan, no systemic study has been undertaken to understand the temporal and spatial changes of forest fragmentation and human impacts on fragmentation in Xishuangbanna to provide conservationists and environmental managers with information on the last remnants of forest fragments and threats to biodiversity. The objectives of the paper is to analyze (1) temporal and spatial pattern of forests change during 1976 and

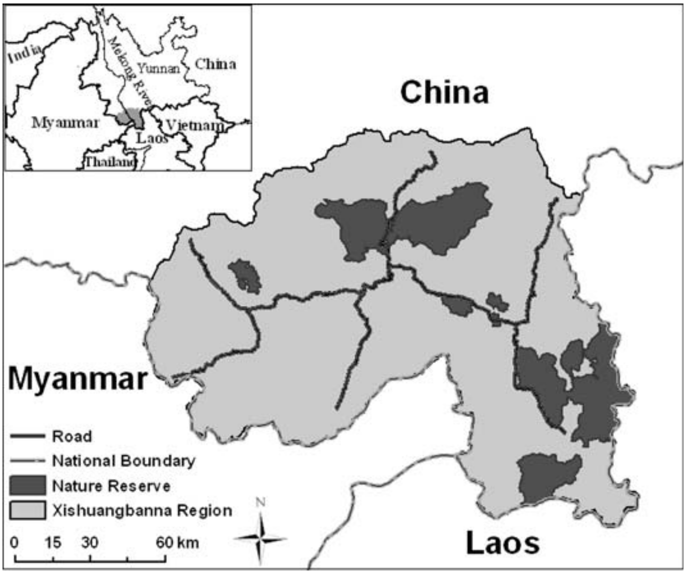

Fig. 1 The location of Xishuangbanna in the southern part of Yunnan province of China

2003 based on three Landsat images and geography information system technology; (2) the patterns of forest fragmentation and the effects of varying edge width in forests by using selected landscape indices; and (3) the degree of forest fragmentation under human interference.

Methods

Study area

Xishuangbanna (21° 08′–22° 36′N, 99° 56′–101° 50′E) located in Yunnan Province, southwest China, covers 19,150 km^2 and borders Laos to the south and Myanmar to the southwest (Fig. 1). The region has mountain-valley topography with the Hengduan Mountains running north-south, and about 95% of the region is covered by mountains and hill. The altitude varies from 2,430 to 475 m above sea level. The climate of this region is influenced by warm-wet air masses from the Indian Ocean in summer, including monsoons, and continental air masses of subtropical origin in winter, resulting in a rainy season from May to October, and a dry season from November to April. The combination of geography and climate in Xishuangbanna has created a transition zone between the flora and fauna of tropical South East Asia and subtropical and temperate China, resulting in the region with the highest biodiversity in China (Zhang and Cao 1995; Cao and Zhang 1997). The five primary forest types in Xishuangbanna are: tropical seasonal rain forest, tropical mountain rain forest, evergreen broad-leaved forest, monsoon forest over limestone, and monsoon forest on river banks (Wu et al. 1987).

Data sources and methods

Land-use/land-cover change was determined using two Landsat Multi Spectral Scanner (MSS) images (24 February 1976—#139/45, and 25 April 1975—#140/45), a Landsat Thematic Mapper (TM) image (2 February 1988—#130/45) and a Landsat Enhanced Thematic Mapper (ETM) image (7 March 2003—#130/45). Two images were used to create the 1976 cover, with information from 1975 used to fill in areas with cloud cover in the 1976 image. All images were acquired during the dry season between February and April. Two land-use maps developed by the Xishuangbanna Department of Land and Resource (Xishuangbanna Land-use Status Map 1982, 1991) and a vegetation map developed by the Xishuangbanna Forestry Bureau (Xishuangbanna Vegetation Distribution Map 1993) were used as references for the classification and accuracy estimation of the MSS and TM images, respectively. Topographic maps (scale = 1:50,000) and digital topographic data with a contour interval of 100 m published by the State Bureau of Surveying and Mapping of China were used to build a digital elevation model (DEM).

The TM satellite images were rectified to Albers Conical Equal Area projection system with a 35-m pixel size. The ETM and MSS images were registered to the TM images using an image-to-image registration technique: rectification RMS errors were <0.5 pixels and <1 pixels, respectively. All non-thermal channels of the TM and ETM images and all channels of the MSS images were used to create class spectral signatures for classification. The images were classified using the supervised maximum likelihood classification method. Training areas for each land-cover class were identified for each image. For the ETM image, training areas were identified in the field during February–March 2003. For the TM and MSS images, training areas were generated from the Department of Land and Resource maps of 1982 and 1991, and the Forestry Bureau's vegetation map of 1993, respectively. We selected large homogeneous areas for the training areas. For each land-use type, we included at least 10 training areas to reflect the variation within a land use due to topography and slope effects. Initially we used the same 15 land-use classes developed by the National Agricultural Zoning Committee (1984). Forests were classified into four classes. It was difficult to distinguish different forest types from the images. The common forest types Xishuangbanna (i.e., tropical seasonal rain forest, mountain rain forest and subtropical evergreen broadleaf forest) were separated based on elevation (Guo et al. 1987). Tropical seasonal rain forest is forested areas with greater than 30% closed canopy dominated by broadleaf trees, and at an altitude less than 800 m. Mountain rain forest is forested areas with greater than 30% closed canopy dominated by broadleaf trees, and at an altitude between 800 and 1,000 m. Subtropical evergreen broadleaf forest is forested areas with greater than 30% closed canopy dominated by broadleaf trees, and at an altitude greater than 1,000 m. Conifers and bamboos could be distinguished based on differences in texture and spectral characteristics. Rubber plantations were easy to classify because the trees are deciduous during the dry season, and most native forest species are evergreen. Shrubland is a common land-use class, but it is often a transition between abandoned agricultural land and forest or plantations. Arable lands included areas of active agriculture, shifting cultivation, grassland, tea gardens, and paddy rice. The land use polygon themes for 1976, 1988 and 2003, obtained from the digital classification of satellite date and subsequent GIS analyses were overlaid and intersected to derive land use/cover changes.

The accuracy of our classification was verified by ground-truthing. Specifically, we compared our classification of the 2003 ETM image with field observations in December 2004. A total of 286 points were verified. In each point, we determined the current land-use cover, determined the location using a global positioning system (GPS), and took a photograph of the site. The field observations were then referenced to the classification to assess the overall accuracy and the accuracy of the different land-use categories. We compared our classification of 1976, 1988 images with two land-use status maps, a vegetation map and topographic maps using 286 points of identical position in 2003. To evaluate the performance of the classification, a confusion matrix was made by comparing the classification results with the reference data based on sample identification (ground information) and some maps (Xishuangbanna Land-use Status Map in 1982, 1991; Xishuangbanna Vegetation Distribution Map in 1993; Topographic maps in 1965). Total accuracy levels of classification of the images for 1976, 1988 and 2003 were 77.3, 86.4, and 87.9%, respectively.

To simplify the analysis of land cover change we used five land cover classes, considering only natural forests and other land use classes. Natural forests included four classes: tropical seasonal rain forest, mountain rain forest, subtropical evergreen broadleaf forest and other forest (including conifer forests and bamboo). Classes for arable lands, shrublands, rubber plantations, water, and urban areas were aggregated to create other land use class in this analysis.

Quantification and comparison of the spatial configuration of forest fragments were conducted based on some landscape indices. Landscape indices were calculated using the software FRAGSTATS Version 3.3 (McGarigal et al. 2002) on the raster data for the coverage for all forest and each forest class in each subset: total Xishuangbanna region, nature reserves and buffers around roads. We used FRAGSTATS to obtain the following characteristics of fragments: the number of patches, mean patch size, largest patch index (LPI, percentage of the landscape comprised by the largest patch) and isolation via Euclidean nearest neighbour distance (NND, average distance to the nearest neighboring fragment of the same patch type). NDD indices were calculated using an 150 m search radius in this study), core and edge habitat area, core area percentage of landscape (CPLAND, the sum of the core areas of each patch of the corresponding patch type divided by total landscape area) and the number of disjunct core area. It is accepted that forest habitat will be more heavily fragmented with increase in the number of patches and isolation and decrease in mean patch size and core area. The indices of core habitat area could be used to evaluate habitat quality, which was important role for interior species survival. The definition of edge width, which was selected to calculate the core and edge habitat area, was arbitrary. Moreover, different species has different response to the edge effect. In order to examine the habitat edge effects, buffers (edge) width of 30, 50, 75, 100, 150, 200 and 300 m in from the perimeter were subtracted sequentially from all forest patches to explore the change of core and edge habitat area and the number of disjunct core area.

To analyze the degree of human interference on forest fragmentation the changes of forest landscape pattern in the total region, within nature reserves and 12.5-km buffers around main road in Xishuangbanna were compared. Digital roads data layer was developed using Topographic maps (scale = 1:50,000) to classify main road constructed in the 1950s within Xishuangbanna. It was reported that the severe land use/land cover change occurred within 12.5-km buffers along road than in total Xishuangbanna region (Cao et al. 2006). The 12.5-km buffers along road were used in our study to assess the states of forest fragments under severe human pressures. In this study fragmentation surrogate variables, such as the number of patch, mean patch size, isolation and the area of core and edge habitat, between total region, protected area and road buffers, were used for comparison. Data regarding other forest classes were excluded from most analyses due to the small sample size in relation to remaining forest classes (Table 1) and mixed forest.

The digital map of forests and other land use was rasterized to a cell size of 30 m when landscape indices were calculated using software FRAGSTATS, a patch being defined as any collection of pixels that touch either at sides or corners, i.e., eight-neighbor clumping method.

Result

Forest landscape dynamics

For the total region forest cover decreased substantially and other land use increased during 1976 and 2003 (Table 1). Forest cover was about 69% in 1976, but it was less than 50% of the studied landscape in 2003. For specific forest classes, subtropical evergreen broadleaf forest was the dominant type in terms of coverage in total forest landscape, followed by mountain rain forest and tropical seasonal rain forest showed intermediate values of these variables and finally the other forest. During the study period, the losses of tropical seasonal rain forest, mountain rain forest, subtropical evergreen broadleaf forest and other forests were 139,576, 103,765, 101,827, and 25,000 ha, respectively.

Within nature reserves, the decrease of forest cover was relatively small, from 88% in 1976 to 84% in 2003 (Table 2). For more specific forest classes, the losses of tropical seasonal rain forest and mountain rain forest area were 2.3% (6,237 ha) and 3.6% (9,743 ha), respectively. However, the area of subtropical evergreen broadleaf forest and other forest increased by 2,754 and 856 ha, respectively.

Within 12.5-km buffers along road the total forests cover decreased significantly from 67% in 1976 to 43% in 2003 (Table 3). The loss of tropical seasonal rain forest amounted to 82,572 ha, from 12% (slightly higher than the percentage of this forest class within total region) in 1976 to 3.3% in 2003 (lower than the percentage within total region). The area of mountain rain forest, subtropical evergreen broadleaf forest and other forest within road buffers all showed substantial decrease.

Fragmentation patterns

One of the basic characteristics of forest fragmentation is the increase in number of patches and the decrease in mean patch size.

Table 1 The change of area, number of patches and mean patch size in Xishuangbanna during 1976 and 2003

	Percent of area (%)			Number of patches			Mean patch size (ha)		
	1976	1988	2003	1976	1988	2003	1976	1988	2003
All forests	69.1	60.3	49.8	6,096	6,724	8,324	217	171	115
TSRF	10.9	8.0	3.6	2,683	3,119	3,894	77	49	18
MRF	15.7	14.7	10.4	2,823	3,381	3,968	107	83	50
SEBF	37.4	34.5	32.0	3,912	4,817	4,187	183	140	146
Other forest	5.1	3.2	3.8	6,661	2,635	3,698	15	23	20
Other land use	30.9	39.7	50.2	16,590	11,959	5,793	36	64	166

TSRF tropical seasonal rain forest, *MRF* mountain rain forest, *SEBF* subtropical evergreen broadleaf forest

Table 2 The change of area, number of patches and mean patch size within nature reserves of Xishuangbanna during 1976 and 2003

	Percent of area (%)			Number of patches			Mean patch size (ha)		
	1976	1988	2003	1976	1988	2003	1976	1988	2003
TSRF	7.9	7.9	5.6	346	347	348	62	62	44
MRF	31.2	30.9	27.6	341	375	452	249	225	167
SEBF	47.8	48.1	48.8	258	244	206	506	537	647
Other forest	1.6	0.6	1.9	549	194	333	8	9	15
Other land use	11.6	12.5	16.1	2,763	2,266	1,460	11	15	30

See Table 1 for abbreviations

Table 3 The change of area, number of patches and mean patch size within 12.5-km buffers around road in Xishuangbanna during 1976 and 2003

	Percent of area (%)			Number of patches			Mean patch size (ha)		
	1976	1988	2003	1976	1988	2003	1976	1988	2003
TSRF	12.0	7.7	3.3	1,450	1,688	2,039	78	43	15
MRF	15.3	13.5	8.5	1,464	1,755	2,219	99	73	36
SEBF	36.3	32.7	28.9	2,127	2,503	2,260	162	123	121
Other forest	3.6	2.2	2.8	3,191	1,321	1,979	11	16	13
Other land use	32.8	43.9	56.6	8,409	5,733	2,556	37	72	209

See Table 1 for abbreviations

For the total region, the number of patches of total forests increased from 6,096 in 1976 to 6,724 in 1988, and then to 8,324 in 2003. However, the mean patch size decreased from 217 ha in 1976 to 171 ha in 1988, and then to 115 ha in 2003 (Table 1). For specific forest classes, the number of patches of tropical seasonal rain forest and mountain rain forest increased significantly and their mean patch size decreased substantially. Particularly, the mean patch size of tropical seasonal rain forest declined from 77 ha in 1976 to 18 ha in 2003 and the number of patches increased from 2,683 in 1976 to 3,894 in 2003. Subtropical evergreen broadleaf forest showed low fragmentations during study periods: the number of patches increased from 3,912 in 1976 to 4,817 in 1988, but decreased to 4,187 in 2003, and mean patch size decreased from 183 ha in 1976 to 140 ha in 1988, and then increased to 146 ha in 2003.

Within nature reserves, there was almost no change in the number of patches of tropical seasonal rain forest, but its mean patch size decreased from 62 ha in 1976 to 44 ha in 2003 (Table 2). The number of patches of mountain rain forest increased from 341 to 452 and the mean patch size decreased from 249 ha to 167 ha from 1976 to 2003. In contrast, the number of patches of subtropical evergreen broadleaf forest decreased from 258 in 1976 to 206 in 2003 and mean patch size increased from 506 ha in 1976 to 647 ha in 2003. The number of patches of other forest also decreased from 549 to 333, and mean patch size increased from 8 to 15 ha.

Within 12.5-km buffers around road buffers there was substantial increase in the number of patches and decrease in the mean patch size in each forest types from 1976 to 2003 except other forest type (see Table 3).

Another characteristic feature of forest fragmentation is that the forest fragments become relatively small. For whole region, 93% of total forest area was concentrated in large patches (≥20,000 ha) and half of the remaining forest occurred in isolated patches of less than 500 ha in 1976, but only 71% of forests area was concentrated in large patches (≥20,000 ha) and forests area in patch size class <20,000 ha increased by 2003 (Fig. 2). For specific forest classes, 51% of tropical seasonal rain forest area was concentrated in large patches (≥5,000 ha) in 1976, especially about 26% of area was in large patches (≥20,000 ha). By 2003, 100% of forest area was concentrated in patches of size less than 5,000 ha and 74% of area was concentrated in isolated patches of less than 500 ha. For mountain rain forest, 32% of area was concentrated in large patches (≥20,000 ha) and 24% of area was occurred in patches size (<500 ha) in 1976. By 2003, no large patches (≥20,000 ha) of forest area remained and 42% of area was concentrated in patches of less than 500 ha. For subtropical evergreen broadleaf forest, about 62% of area was concentrated in large patches (≥20,000 ha) in 1976, but only 48% was remained in large patches (≥20,000 ha) in 2003.

The size of the largest total forests patch decreased from 26% of the total region in 1976 to 12% in 2003. The large forest patches disturbed was main cause of forest fragmentation.

The substantial reduction of forest cover also was accompanied with isolation of the fragments from 1976 to 2003 (Fig. 3). The mean nearest neighbour distance (NDD) of total forests decreased from 144 m in 1976 to 194 m in 2003. For specific forest classes, the NDD of tropical seasonal rain forest increased significantly from 159 m in 1976 to 253 m in 2003, mountain rain forest from 176 to 230 m, and subtropical evergreen broadleaf forest from 165 to 199 m.

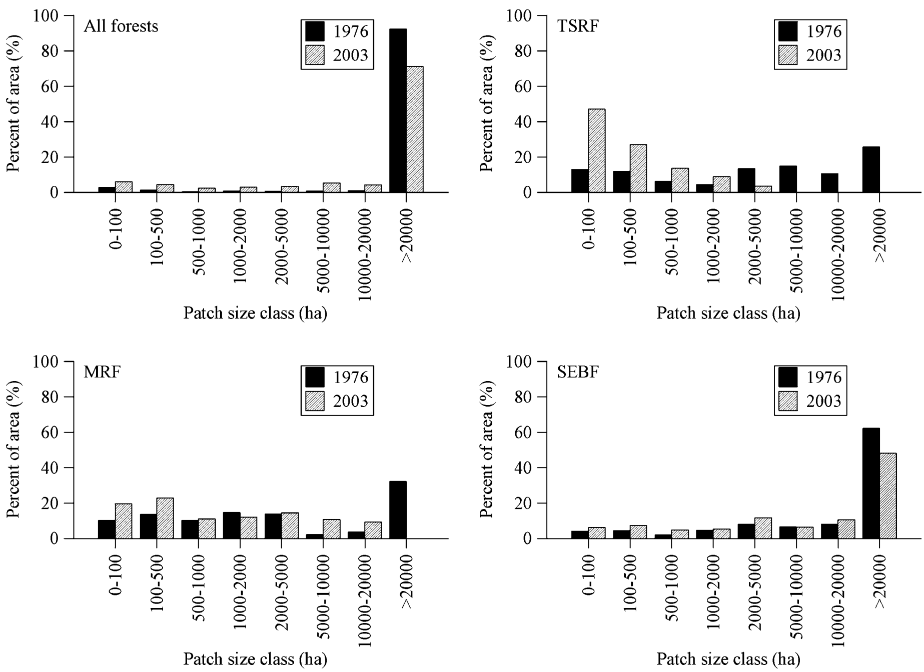

Fig. 2 The percent of area at different patch size across the total Xishuangbanna region from 1976 to 2003. See Table 1 for abbreviations

Effect of edge width on forest core habitat area

The core habitat area of each forest types substantially decreased as the edge width increased, but the number of disjunct core areas increased within edge width ≤50 m, then decreased at edge width >50 m (Fig. 4). The largest number of disjunct core areas at narrow edge zone was correlated with irregular or amoeboid forest patch shape. At edge width ≤50 m, more than one core habitat area might occur within one patch. With increasing edge width, no core habitat area remained in irregular or amoeboid shaped patches. For tropical seasonal rain forest and mountain rain forest, the number of disjunct core areas in 2003 was larger than that in 1976 at narrow edge width (<50 m), but with edge width increasing it showed reverse change. The increasing number of disjunct core areas within narrow width edge zones in 2003 was associated with the increase of small patches of these two-forest types in this time (Fig. 2). For subtropical evergreen broadleaf forest, the number of disjunct core areas in 2003 was substantially smaller than that in 1976 at any edge width zones.

The core habitat area of each forest classes declined consistently over time at different edge width (Figs. 4, 5). For the edge buffer of 75 m, a decline of core habitat area (CPLAND) was observed from 7% in 1976 to 2% in 2003 in tropical seasonal rain forest, from 11 to 7% in mountain rain forest, and from 29 to 25% in subtropical evergreen broadleaf forest. In particular, almost no core habitat area remained in tropical seasonal rain forest at the edge width of 300 m in 2003. At edge width 75 m, the area of edge habitat of tropical seasonal rain forest has already exceeded core habitat area (the difference between the area of edge habitat and core habitat was 1,500 ha) in 2003, but at the same edge width, the area of edge habitat did not exceed core habitat in 1976 (Fig. 5). At edge width of 100 m of the mountain rain forest, the area of core habitat was larger than edge area in 1976 (the difference was 38,000 ha), but the area of core was almost equal to the area of edge habitat in 2003 (Fig. 5). For subtropical evergreen broadleaf forest the area of edge habitat exceeded core area (the difference was 17,000 ha) at edge width of 200 m in 1976, but the area of edge habitat did not exceed the area of core habitat in 2003 (the difference was 30,000 ha) at this edge width (Fig. 5), because the mean patch size of this forest type increased in 2003.

Comparing the core habitat area, the percentage of core habitat area inside nature reserves was generally larger than that the whole region and road buffers at any edge width and any time except CPLAND of tropical seasonal rain forest in 1976 (Fig. 6). The

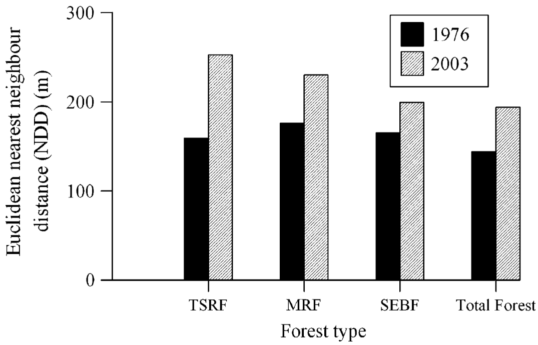

Fig. 3 Change of isolation of different forest type from 1976 to 2003. See Table 1 for abbreviations

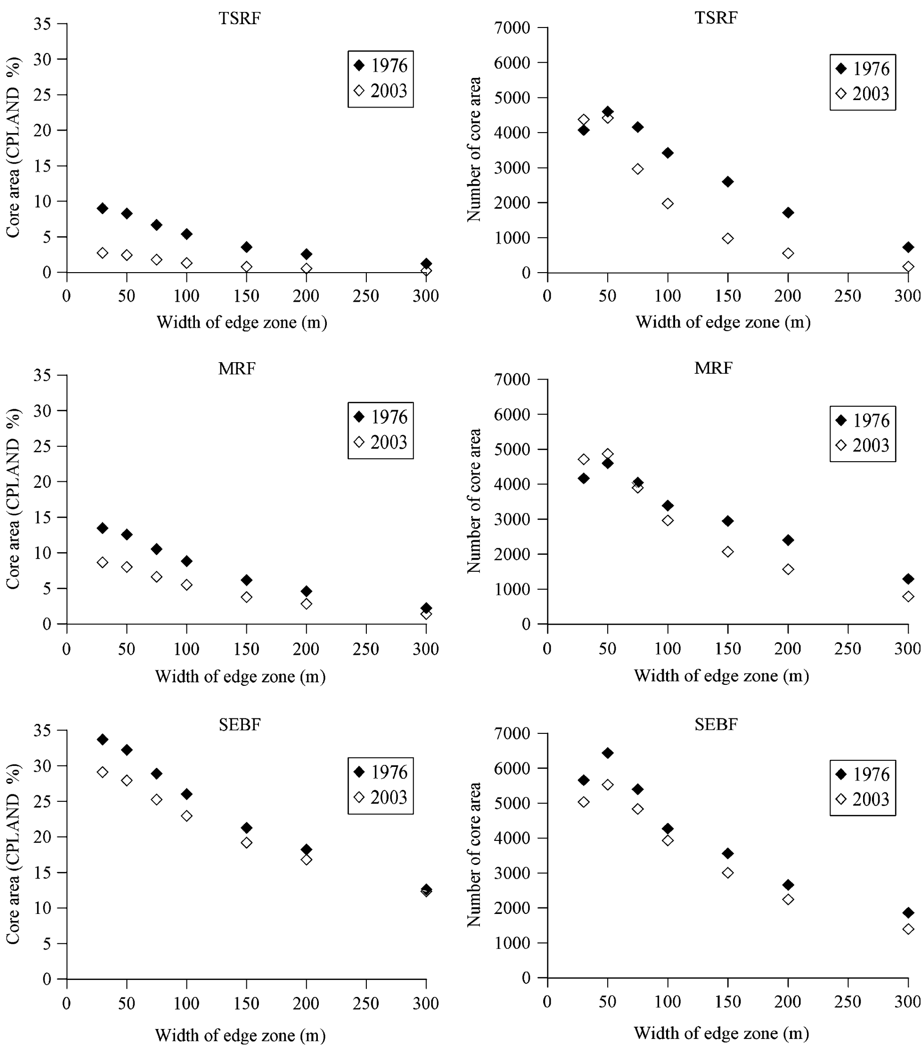

Fig 4 Change of core areas and the number of disjunct core areas at different edge width across the total Xishuangbanna area in 1976 and 2003. See Table 1 for abbreviations

percentage of core habitat area of each forest class within 12.5-km buffers around road was the smallest at different edge width. Particularly, being affected by human activities, no core area remained in tropical seasonal rain forest at edge width 300 m in 2003 within road buffers (Fig. 6). In 1976, the CPLAND of tropical seasonal rain forest was smaller within nature reserves than that within whole region and road buffer, because most reserves were located at relatively higher altitude where mountain rain forest and subtropical evergreen broadleaf forest were dominant types. In 1976, a large forest cover still remained across whole region and tropical seasonal rain forest disturbance was also slight in the total region, nature reserves and road buffers. However, by 2003, a large area of tropical rain forest lost outside reserves, but the loss of tropical rain forest within the reserves was not substantial.

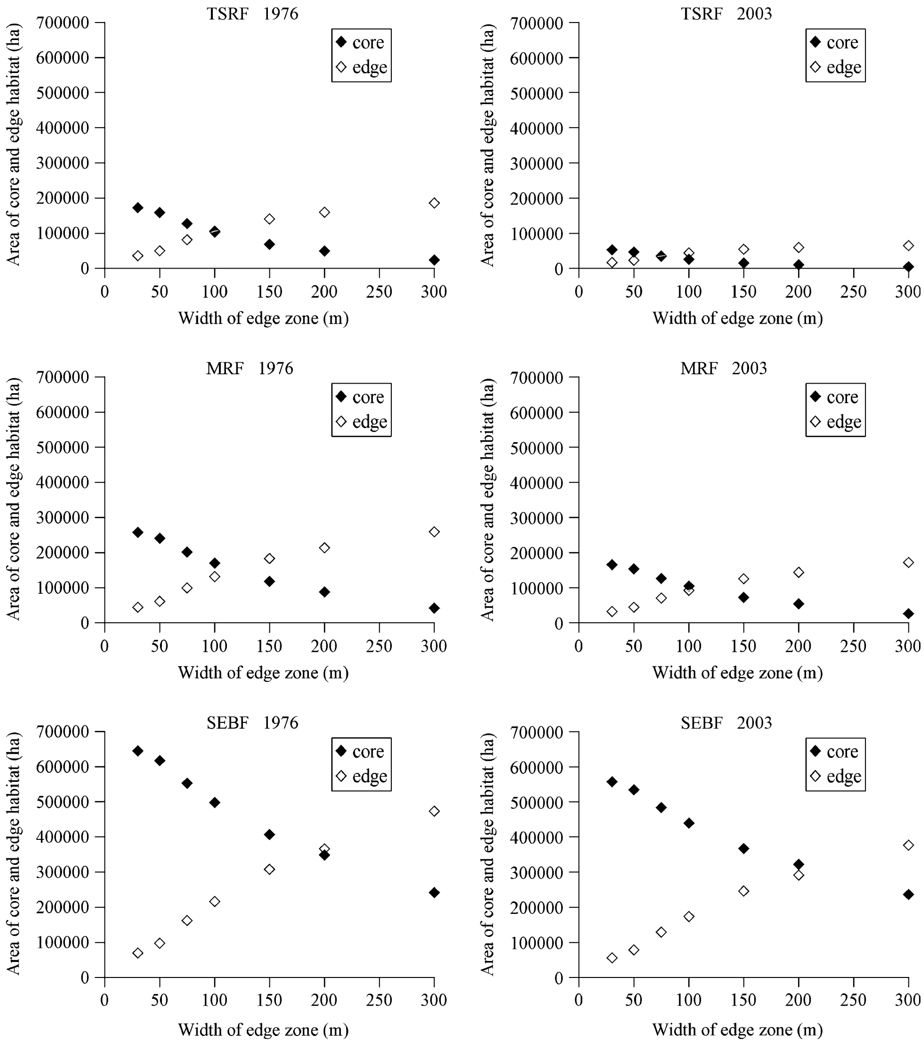

Fig. 5 The relationship between core and edge habitat area, and edge width modeled for different forest type across total Xishuangbanna area. See Table 1 for abbreviations

Discussion

The causes of forest fragmentation

The agriculture lands expansion and natural forest area loss were the main characteristics of land use/land cover change from 1976 to 2003 in Xishuangbanna. The practice of shifting cultivation caused by some polices was the main reason of deforestation from 1976 and 1988 (Xu et al. 2005). Rubber plantations in tropical seasonal rain forest, mountain rain forest and shifting cultivation lands at low altitude were the major causes of deforestation from 1988 to 2003 (Liu et al. 2006, 2007). The growth of human population and the lack of alternative economic opportunities in Xishuangbanna drove the demand for

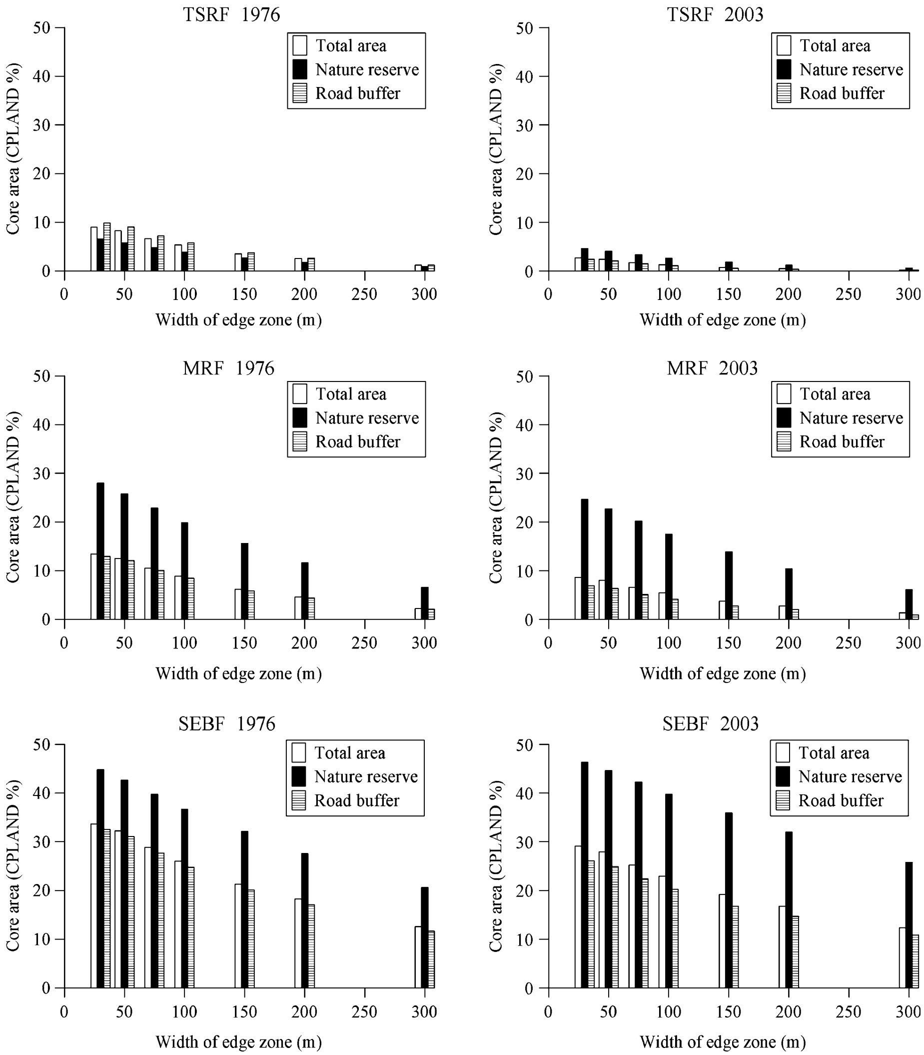

Fig. 6 Core area percentage of landscape in total area, nature reserves and 12.5-km buffers around road in 1976 and 2003

arable lands for livelihoods and illegal clearing of forest on high attitude or steep slope (Li et al. 2007).

The economic structure and natural geographic topography influence landscape cover change, size distribution and average fragment size of forest classes. Our study showed that subtropical evergreen broadleaf forest remained with large area of interior, but the remaining fragments of tropical seasonal rain forest and mountain rain forest were small and had a high level of degradation and fragmentation. A large area of tropical rain forests distributed at lower altitude region where human activities were concentrated was utilized as arable lands. In addition, these areas are exactly suitable for rubber tree growth. Increase in the demand for rubber products stimulated the rubber plantations expansion quickly in past decades, and convert a large area of tropical rain forest and shifting cultivation into rubber plantation (Li et al. 2007).

Subtropical evergreen broadleaf forests are located in the high mountains with lower degree of humane activities. Furthermore, compared to tropical rain forest, subtropical evergreen broadleaf forest can easily regenerate. Ou et al. (1997) found that restoration of tropical rain forest was always slower than that of subtropical evergreen broadleaf forest in Xishuangbanna and the disturbed primary tropical rain forests would be potentially replaced by subtropical evergreen broadleaf forest in this region. This implicates that the actual rates of tropical rain forest loss may be even higher, because proportion of tropical rain forests have been degraded to other forest types.

The pattern of fragmentation and implication

Following the general pattern in most tropical rain forest region (Ranta et al. 1998; Cayuela et al. 2006), the general trend of forest fragments was shrinking forest fragment size, increasing number of fragments and isolation of the remaining habitat patches in Xishuangbanna. Whereas the remaining large and continuous forest cover decreased drastically since 1976. Deforestation led to an increase in the number of small forest patches, edge habitat area and isolation, and a decrease in mean patch size and core habitat area from 1976 to 2003. Our results showed differences among forest classes in terms of landscape cover, size distribution and average fragment size. Subtropical evergreen broad leaf forests show a higher internal patchiness mainly because of their comparatively larger area. Fragment size distribution of tropical seasonal rain forest, which has almost the same forest profile and physiognomic characteristics as equatorial lowland rain forest and the richest biodiversity (Zhu 2006), is strongly skewed towards small values indicating heavy fragmentation. At the narrow edge width, such as 75 m, the edge habitat area of tropical seasonal rain forest has exceeded core habitat in Xishuangbanna (Fig. 5). The edge effect in tropical seasonal rain forest was similar in magnitude to that in the Atlantic rain forest of Pernambuco that the edge habitat area exceeded core habitat at the edge width of 60 m (Ranta et al. 1998). Our results also showed 100% of tropical seasonal rain forest area was concentrated in patch size (<5,000 ha) and about 86% of area was concentrated in isolated patches of less than 1,000 ha in 2003 in Xishuangbanna (Fig. 2). The current state of tropical rain forest fragments may be inhospitable to some species (such as Asian elephant) migration and survival. The Asian elephant preferred to select habitat with an altitude less than 1,000 m, especially below 800 m (Feng and Zhang. 2005; Yang et al. 2006), which is exactly the area of heavy forest fragmentation in Xishuangbanna. Home range sizes of Asian elephants normally range between 3,400 and 80,000 ha (Stuwe et al. 1998). It was reported that about 1,000 ha fragments with less disturbed are needed for their survival based on a research on the main habitat the five Asian elephant herd frequently used in Simao, China (Zhang and Wang 2003). The tropical rain forest fragments that are too small to support such largest herbivore are the most important factor why the human-elephant conflicts have become more serious in Xishuangbanna (Zhang and Wang 2003). The tropical rain forest habitat reduction and fragmentation force the Asian elephant to come out of nature forest to forage and breed on agricultural field (Feng and Zhang. 2005). Careful consideration is needed when some of consequences of fragmentation are assessed. Although fragments will induce some species' abundance decline and others becoming hyperabundant (Laurance et al. 2002), the empirical evidence suggests that species' abundance in fragments differ from intact forest and the large forest area has high species richness capacity (Brooks et al. 1999, 2002; Debinski and Holt 2000; Laurance et al. 2002;

DeFries et al. 2005). By comparison of the plant species on fragmented forest and primary forest in Xishuangbanna, it was reported that the species richness was low in fragmented forest and about 22.4% species was lost or replaced by other species after 30-year isolation on fragment of tropical rain forest in a Dai's holy hill (Zhu et al. 2000, 2001).

Disturbed rainforests with forest fragmentation differ greatly from intact forest in their biophysical characteristics (Laurance 2004). Furthermore, forest-size category and adjacent land use were the most important factors in determining species composition and the distance of edge effects of the species disturbed or penetrated (Guirado et al. 2006; Storch et al. 2005). It was found that the most striking edge effect (such as tree mortality and loss of aboveground forest biomass) occurred within 100 m of forest edges and wind or fires damage to forests may penetrate 300 m or more into tropical forest remnant (Laurance et al. 1997, 1998, 2002, 2004; Debinski and Holt 2000; Flaspophler et al. 2001). Moreover, when fragments smaller than 100–400 ha in area, edge effects should have a rapidly increasing impact on forest dynamics (Laurance et al. 1998). Our results indicate that 63% of remaining tropical seasonal rain forest and 47% of remaining mountain rain forest in Xishuangbanna was susceptible to edge-association damage and tree mortality in 2003. Ninety-four percent of the remaining tropical seasonal rain forest and 87% of the mountain rain forest in Xishuangbanna may be subjected to microclimate edge effects in 2003. Study on the edge effects on soil seed bank and understory vegetation in tropical seasonal rain forests in Xishuangbanna showed the invasion of a majority of non-forest species in understory vegetation lags behind the accumulation of their seeds in soil banks in forest edge zone, since the conditions are not appropriate for non-species' establishment (Lin and Cao 2009). This implies potential edges created by edge influence within the tropical rain forest could promote non-forest species to establish further into the forest. The analysis of spatial and temporal change of edge and core habitat area at landscape level can provide conservationists and environmental managers with more important information on the current state of forest fragmentation and degradation. Moreover, it will be usefulness to establish conservation planning on managing habitats. However, edge effects in forest fragments are significantly influenced by the structure of surrounding vegetation. It was found that fragments surrounded by regrowth forest are somewhat buffered from damaging winds and harsh external microclimates, and suffer lower edge-related tree mortality than do those encircled by cattle pastures (Mesquita et al. 1999). We think that the concrete distance and magnitude of edge effects also need to be validated with the more field investigation data or model simulation considering the biota and abiotic impacts (Zheng and Chen 2000).

Isolation of fragments is one of the main factors affecting the colonization possibilities of species (Ranta et al. 1998; Tischendorf and Fahring 2000; Piessens et al. 2005). Meyer and Kalko 2008 (in press) studied land-bridge island in Panama and found that island (fragment) isolation rather than area was linked to patterns of nestedness in bat assemblages. In heathland area of Flanders, the incidence of almost three quarters of the species was influenced by fragmentation and isolation was the most important factors determining their presence or absence in a heathland patch (Piessens et al. 2005). The large tracts of arable lands buffer the isolated sub-reserves within Xishuangbana region (Fig. 1) has influenced the Asian elephant's intercommunication (Liu et al. 2008). The increasing degree of isolation of forest fragments in the past decades (Fig. 3) may suggest a vast loss of forest connectivity and influence the more species dynamics.

The large tracts of forest remained inside nature reserves and had lower degree of fragmentation in Xishuangbanna. The density of patches within nature reserves was lower than that within whole Xishuangbanna area and 12.5-km buffers along road, but mean patch size was the biggest (Table 4). The larger core habitat areas were remained as the forests cover inside nature reserve was comparatively stable (Fig. 6). Debinski and Holt (2000) demonstrated that reserves could minimize the edge-to-area ratio to maximize the effective core habitat area of the reserve. However, the tropical rain forest inside protective area has moderate degree of fragmentation with mean patch size reduction and vulnerable to edge effects due to human disturbance from 1976 to 2003. Moreover, the new highway is just across the nature reserves in Xishuangbanna (Fig. 1). The edge effects can reduce reserve effectiveness (Revilla et al. 2001). Although there is no valid data about mortality of animal near the reserves border in Xishuangbanna, human-related mortality on animals has been reported inside protected areas in other tropical region, especially near their borders and roads (Revilla et al. 2001). Defries et al. (2005) found that the presence of forest habitat outside the administrative boundaries of most protected areas enhanced their species richness capacity beyond the forest habitat within the boundaries alone. It is necessary to manage human activities inside and outside protected areas so as to reduce edge effects and to protect the forest inside and outside reserves. Under the impact of human activities, forest fragmentation was more severe outside reserves, especially the area around roads in the past decades in Xishuangbanna. The road was the concentration of the human activities and the common spatial arrangement of settled areas was along roads. The road would create barriers for species dispersal or as corridor within landscape for exotic species invasion (Bhattacharya et al. 2003; Gelbard and Belnap 2003). How to manage the roads, especially the roads traverse the nature reserves, has important conservation implication for native biodiversity. In our study, we only considered the effects of main tar roads on forest fragmentation. There are many dirt and gravel roads in this region that would aggravate forest fragmentation (McGarigal et al. 2001). It was found that roads had a much greater impact on landscape structure than logging and road edges may persist longer than natural patch edge or those created by clearance (Reed et al. 1996; McGarigal et al. 2001).

The remaining large forest fragments are important protection places as they could contain large core habitat area avoiding influences of edge effect. The small fragments also have important value for most species survival (Piessens et al. 2005). For example, the research on fragmentation impacts on insectivorous bat species reported that fragments >300 ha contribute substantially to landscape-level bat diversity, but the smaller fragments also have substantial value for bat diversity (Struebig et al. 2008). The area of high-diversity tropical season rain forest has almost been eliminated, so remaining fragments

Table 4 Comparison of the patches in different study area in 2003

	TSRF	MRF	SEBF
Total study area			
Patch density (/ha)	0.2033	0.2072	0.2186
Mean patches size (ha)	18	50	146
Nature reserves			
Patch density (/ha)	0.1276	0.1657	0.0755
Mean patches size (ha)	44	167	647
12.5-km buffers around road			
Patch density (/ha)	0.2157	0.2347	0.2391
Mean patches size (ha)	15	36	121

See Table 1 for abbreviations

was priority to be protected effectively in Xishuangbanna. For conservation purposes, the networks of forest fragments connected by corridors and stepping stone fragments are important for species immigrants, genetic exchange, effective population sizes maintain, and so on (Haddad and Baum 1999; Mech and Hallett 2001; Orrock et al. 2003). Based on forest fragmentation pattern, the corridors could be built using river riparian forest, or remaining small forest fragments as stepping stone, and keeping proportional area of forest regeneration around the remaining forest fragments. We also suggest redefining some of the protected areas of the nature reserves system based on the spatial configuration pattern of native forest fragments to include surrounding areas of natural ecosystem before they are further degraded. Especially, remaining smaller tropical rain forest patches situated in small open patches in forests or along roadside are much more endangered. Management of the sensitive edge habitat is also important to ameliorate the effects of forest shrinkage. In case of small forest fragments or patches with amoeboid shape, building hospitable vegetation (i.e., dense plantations) buffer forest fragments is necessary to alleviate the edge effects and forest fragments isolation (Denyer et al. 2006). The large tracts of rubber plantations are increasing adjacent to tropical rain forest in Xishuangbanna (Li et al. 2007). The monoculture rubber plantation could not resemble to the nature primary forest, but agroforestry (i.e., rubber + tea) has higher biodiversity than monoculture rubber plantation (Liu et al. 1998). The multilayer plantation built could provide valuable microclimate buffering during the day, principally due to their effect in reducing light and temperate to interior-like conditions at native forest edges (Ma et al. 1998; Denyer et al. 2006).

Biodiversity conservation is complicated and systemic project, which need collaboration between government agencies, society and researchers. Our results quantificationally evaluated the forest cover change and the pattern of forest fragmentation in Xishuangbanna and provided critical information for biodiversity conservation and local land use management, especially for the initiative establishment program of local biodiversity conservation corridor.

Acknowledgments This work was funded by the National Natural Science Foundation of China (30770385 and 30770368). We thank Z. F. Guo and Z. W. Cao who assisted with early stages of geographic information system analysis and fieldworks. Y. H. Liu provided invaluable assistance with data collection. We offer special thanks to Y. Q. Zhang for his helpful comments on the manuscript. The authors are very grateful to the anonymous reviewer for providing constructive comments and suggestions that improved this manuscript.

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