Taylor & Francis
Taylor & Francis Group

ARTICLE

OPEN ACCESS

Check for updates

Simulating land-use change and its effect on biodiversity conservation in a watershed in northwest China

Youjia Liang[a] and Lijun Liu[b]

[a]School of Resources and Environmental Engineering, Wuhan University of Technology, Wuhan, China; [b]School of Navigation, Wuhan University of Technology, Wuhan, China

ABSTRACT

Introduction: Land-use management strategies play a major role in biodiversity change. In many parts of the world, local governments are under increasing pressure to regulate human activity to mitigate negative impacts on ecosystems.

Outcomes/other: This study aimed to analyze the effects of different land-use patterns on biodiversity change across a typical artificial desert watershed. We first analyzed land-cover change based on past and future management scenarios in a watershed spanning Gaotai, Linze, and Ganzhou counties in northwest China. We then analyzed the effect of different land-use patterns on biodiversity change in the watershed. We found that the crucial land-cover changes are likely to occur in the wetland reserves and areas established for the Grain for Green Project around the oases, and such changes could affect biodiversity throughout the entire watershed landscape.

Discussion: The use of spatial analysis to illustrate explicit changes in ecosystems is useful in fostering biodiversity awareness and the need for decision-making at different scales.

Conclusion: Thus, these findings indicate that land-use management strategies for the middle and southeast parts of the watershed are particularly important for future management of biodiversity and the integrated ecosystem services of the entire watershed landscape.

Background

The importance of biodiversity to human activities has been widely recognized (Butchart et al., 2010; Rands et al., 2010; Steffen et al., 2009). In China, biodiversity continues to decline in spite of efforts by the Chinese government to manage different threats and prevent ecosystem degradation (Xiao et al., 2005; Xie et al., 2005; Xie et al., 2010). Specifically, some watersheds in northwest China have suffered large extinctions of species as a result of intense human activity and rapid economic development over the last 20 years. Rapid economic development has led to reductions in strict land-use management in watershed landscapes, which has resulted in changes in biodiversity and ecosystem functions. The main identified threats to biodiversity in the watershed landscape include land fragmentation, degradation of specific habitats or land-cover types (Baral et al., 2014), unsustainable use of natural resources (e.g., rapid decline of forest and grassland), inappropriate cropping systems, climate change, and natural loss. Thus, predicting land-use change and its effects on biodiversity conservation is crucial in regional land-use management and planning (Geneletti, 2013).

With rapid expansion of agricultural activity, the role of alternative land-use management strategies is also increasing on production landscapes for conserving biodiversity and providing ecosystem services (e.g., Kandziora, Burkhard, and Müller, 2013; Wilson et al., 2010; Foley et al., 2005). The impact of future land-use change on biodiversity conservation at the local scale has been addressed in recent case studies (e.g., Minin et al., 2017; Goldstein et al., 2012). In addition, assessment of GIS-based techniques and spatial explicit models for biodiversity change is essential to aid decision-making and planning in landscapes dominated by intensive human activities (Xuan et al., 2017; Nelson et al., 2010; Eigenbrod et al., 2009; Brooks et al., 2006).

This study proposes a method for integrating land-use change into the quantification of biodiversity conservation. To illustrate usefulness and effectiveness of the integrated modeling method, the Zhangye watershed located in the middle basin of Heihe River in northwest China is used as a case study. The aim of this study was to analyze the quantitative effects of some alternative landscape management practices on biodiversity in the watershed. The study mainly has three specific objectives: (1) to simulate and validate land-use change

patterns across the watershed landscape based on CLUE-S (Conversion of Land Use and its Effects at Small regional extent) model from 2000 to 2009; (2) to develop and use three landscape management scenarios to simulate the future land-use change, which indicate different combined strategies and policies for land demand–supply balance from 2010 to 2014 in the watershed; and (3) to quantify and map changes in biodiversity using the Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) biodiversity model that caused by land-use change from 2000 to 2009 and predicted future land-use change under the potential land-use management strategies, respectively. Finally, the selected biodiversity indicators were compared to discuss the alternative land-use change patterns in the study area. Our findings can be used to support spatial natural resource planning in this watershed landscape.

Methods

Study area

The Zhangye watershed including three cities of Ganzhou, Linze, and Gaotai is located in the middle basin of Heihe River, northwest China, and the study is between 98°57′–100°52′E and 38°32′–39°42′N (Figure 1). The total area of the watershed is 1.13×10^4 km². Annual evaporation of the watershed is 1000–2000 mm, and mean annual precipitation is 62–156 mm (Zhao, Liu, and Zhang, 2010). There has

been a rapid period of agricultural development in the watershed during the past 50 years. The unsustainable agricultural expansion with the fact of water supply barely meets the overall demand is a typical problem in such an arid/semiarid region in northwest China. From 2000 to 2009, the total population of this watershed increased by 77.94%, from 352,380 to 627,022, and the gross domestic product increased by 95.99%, from 4.06×10^{10} RMB to 7.96×10^{10} RMB (ZSB 2000; ZSB 2009). The rapidly increasing population pressure and economic growth have resulted in extensive exploitation of water resources, agricultural expansion, and ecosystem degradation. Now, the most typical land-use pattern in the study area is urbanization and cultivated land use with large-scale intensified agricultural activities, and it also include main land-cover types of water, forest, grassland, and unused land. As a result, the temporal and spatial assessment of biodiversity is of great importance for future sustainable landscape policymaking and watershed management.

Data

Different datasets were combined to quantify and map biodiversity in the watershed. Land use/cover data for the research were comprised of four different remotely sensed images recorded on: 16 August 1995, 21 August 2000 (Landsat-5 Thematic Mapper image, 30 m spatial resolution, http://land

Figure 1. Location of the artificial desert watershed.

Table 1. Input data types and description for the InVEST biodiversity model.

Input data	Description
LUCC map	A raster map with a numeric LUCC code for each cell
Threat table	A table of different threats considered for analysis, e.g., agriculture, desert, and roads
Threat sources	Raster files of the distribution and intensity of each threat
Accessibility to sources of degradation	A polygon shape file containing data on protected areas, which provide relative barriers against threats. Conservation areas and protected lands were considered sites with minimum accessibility and were assigned a threat level of 0, while polygons with maximum accessibility were assigned 1 (Himlal et al., 2014; Tallis et al., 2013; Polasky et al., 2011)
Sensitivity of habitat types to each threats	A table of LUCC types. Sensitivity values range from 0 to 1 where 0 represents no sensitivity to a threat and 1 represents the greatest sensitivity. Sensitivity scores were determined from the literature and expert knowledge (Himlal et al., 2014; Tallis et al., 2013; Polasky et al., 2011)
Half-saturation constant	The InVEST model uses a half-saturation curve to convert habitat degradation scores to habitat quality values. An inverse relationship between the degradation score and its habitat quality value is determined by this half-saturation constant. The half-saturation constant used was equal to the grid cell degradation score that returns a pixel habitat quality value of 0.5 (Tallis et al., 2013)

sat.datamirror.csdb.cn), 23 November 2005, and 9 September 2009 (SPOT-5, 20 m, http://westdc.west gis.ac.cn/). In addition, other raster data were collected which comprise digital elevation model (100 m), road map and boundary map (China basic geographic information database, http://nfgis. nsdi.gov.cn/), wetland boundary map, and ancillary data of the study area from local government. All of these images and other raster data were geometrically corrected and geocoded to the Universal Transverse Mercator coordinate system using an existing reference topographic map. For the raster data, a cubic convolution algorithm was used to data preprocessing and the transformation had a root mean square (RMS) error between 0.85 and 1 (Liang and Liu, 2014), indicating that the accuracy of these images was less than a pixel. The input data variables and resolution were held constant for all of the modeling approaches. The data input layers were then resampled to a common scale of 100-m spatial resolution using the Nearest Neighbor resampling methods in ArcGIS 10. This is generally an appropriate spatial resolution in the majority of land-use model applications, which are in the range of 30–100 m (Guan and Clarke, 2010). Detailed input data for the CLUE-S and InVEST biodiversity model in this study are outlined in Table 1.

Simulation method

In this study, we used the InVEST model (Tallis et al., 2013) to simulate biodiversity change from 2000 to 2009 and under the two future land management scenarios. The model uses habitat quality (Polasky et al., 2011) as a proxy for biodiversity assessment. Generally, degradation of habitat quality is caused by the intensity of nearby land-use expansion in relation to intensive human activities. At the pixel scale, an exponential decay function can be used to describe the impact i_{rxy} of threat r from pixel cell y on habitat in cell x:

$$i_{rxy} = \exp\left(-\left(\frac{2.99}{d_{r\max}}\right)d_{xy}\right) \qquad (1)$$

where d_{xy} is the linear distance between pixel cells x and y (km), and $d_{r\max}$ is the maximum effective distance of threat r reach across space (km). Thus, a pixel cell threat level is translated into a habitat quality using the total threat level and a half saturation function.

$$D_{xj} = \sum_{r=1}^{R}\sum_{y=1}^{Y_r}\left(\frac{w_r}{\sum_{r=1}^{R}w_r}\right)r_y i_{rxy}\beta_x S_{jr} \qquad (2)$$

$$Q_{xj} = H_j \times \left(1 - \left(\frac{D_{xj}^{2.5}}{D_{xj}^{2.5}+k^{2.5}}\right)\right) \qquad (3)$$

where D_{xj} is the total threat level in grid cell x with land-use type j; y is all grid cells on r raster map; Y_r is the set of grid cells on r raster map; threat weight w_r is the relative destructiveness of a degradation source to all habitats; and r_y is raster map r. β_x is the level of accessibility in grid cell x, where 1 indicates complete accessibility; the values of relative sensitivity S_{jr} of each habitat type to each threat (*L_threat x; crp* refers to cropland, *rr* to rural residential, *urb* to urban, *rot* to rotation forestry, *prds* to primary roads, *srds* to secondary roads, and *lrds* to light roads) range from 0 to 1, where 1 represents high sensitivity to a threat and 0 represents no sensitivity to a threat. Q_{xj} is habitat quality value of land use type j; H_j is a habitat quality score that ranges from 0 to 1, where non-habitat land-use types are given a score of 0 and perfect habitat classes were scored 1; and the half-saturation constant k is 0.5 (Tallis et al., 2013). The weight of threat *crp, rr, urb, rot, prds, srds*, and *lrds* was 8, 5, 7.5, 6, 3, 1, and 0.5, respectively. The maximum effective distance of threat *crp, rr, urb, rot, prds, srds*, and *lrds* was 0.7, 0.6, 0.8, 0.5, 1, 0.7, and 0.5, respectively. The other detailed input data for the InVEST biodiversity model in this study are outlined in Table 2.

Land management scenarios in the watershed

The land use/cover map provides an environment for exploring the consequences of different land management policies (Liang, Liu, and Huang, 2017; Liang

Table 2. Habitat quality scores and sensitivity of different habitat types to threat sources.

LUCC	Coding	H_j	β_x	L_crp	L_rr	L_urb	L_rot	L_prds	L_srds	L_lrds
Forest	21	0.6	1	0.65	0.8	0.6	0.4	0.5	0.4	0
Spinney	22	1	1	0.8	0.85	1	0.6	0.8	0.7	0.6
Open woodland	23	1	1	0.7	0.75	0.9	0.5	0.7	0.6	0.5
Other woodland	24	1	1	0.3	0.35	0.5	0.1	0.3	0.2	0.1
High-coverage grassland	31	0.3	1	0.35	0.5	0.3	0.1	0.2	0.1	0
Middle-coverage grassland	32	0.4	1	0.45	0.6	0.4	0.2	0.3	0.2	0
Low-coverage grassland	33	1	1	0.3	0.35	0.5	0.1	0.3	0.2	0.1
Canal	41	0.7	0.5	0.75	0.9	0.7	0.5	0.6	0.5	0
Lake	42	1	0.2	0.7	0.75	0.9	0.5	0.7	0.6	0.5
Glaciers and permanent snow	43	1	1	0.7	0.75	0.9	0.5	0.7	0.6	0.5
Shallow	46	1	1	0.3	0.35	0.5	0.1	0.3	0.2	0.1
Urban areas	51	1	1	0.4	0.45	0.6	0.2	0.4	0.3	0.2
Rural areas	52	0.3	1	0.35	0.5	0.3	0.1	0.2	0.1	0
Other construction areas	53	0	1	0	0	0	0	0	0	0
Desert	61	1	1	0.3	0.35	0.5	0.1	0.3	0.2	0.1
Gobi	62	0	1	0	0	0	0	0	0	0
Saline-alkali fields	63	1	1	0.3	0.35	0.5	0.1	0.3	0.2	0.1
Marsh	64	0.7	0.8	0.75	0.9	0.7	0.5	0.6	0.5	0
Bare areas	65	0	1	0	0	0	0	0	0	0
Rock	66	0	1	0	0	0	0	0	0	0
Paddy field	111	1	1	0.3	0.35	0.5	0.1	0.3	0.2	0.1
Mountain cropland	121	1	1	0.3	0.35	0.5	0.1	0.3	0.2	0.1
Plains cropland	123	1	1	0.3	0.35	0.5	0.1	0.3	0.2	0.1

and Liu, 2014). We prepared land-use maps for 2000 and 2009 and then compared the maps for a biodiversity change analysis.

To provide a context that is understandable to regional managers, we mapped the watershed based on the current land use/cover map of 2009 using two simple scenarios: (1) moderate protection (SP1), where change of wetland reserves (the National Wetland Reserve of Heihe River, 973.68 km^2) was limited according to environmental considerations; and (2) strict protection (SP2), which simulated strict protection of both wetland reserves and Grain for Green Project areas (172.64 km^2) in the watershed. The management scenarios were rooted in plans derived from existing local government planning and policy making. For example, in 2011, the Grain for Green Project areas and wetland reserves in the watershed were placed under protection in the regional ecological conservation plan (Liang and Liu, 2014). The two scenarios considered in this study provide a general guide for local government managers, as well as for a larger audience of different groups and stakeholders involved in economic development and ecosystem conservation in the watershed. The goal of the analysis is to help policy makers and land managers to understand the trade-offs of different land management strategies and to appreciate some of the ecological outcomes of different development policies.

Results

Changes of land use/cover

The different land use/cover maps for 2000, 2009, SP1, and SP2 were used to assess the change of biodiversity in the study area. The land use/cover coding in Figure 2 corresponds to the numbers 21–123 in Table 2. Based on the land-use change analysis, the change of each land-use type in 2000 and 2009 is re shown in Table 3. The gobi, desert, bare areas, and rock cover types were the dominant land coverage in each year. However, the proportion of these land-cover types declined from about 61.3% in 2000 to 53.87% in 2009. Thus, the proportion of watershed areas in the whole artificial desert watershed landscape increased from about 38.7% in 2000 to 46.13% in 2009. Specifically, the proportion of grassland increased from about 10.3% in 2000 to 20.21% in 2009. Meanwhile, forest and other woodland increased from about 0.75% in 2000 to 1.38% in 2009, and the proportion of cropland declined from about 20.12% in 2000 to 15% in 2009. In general, the total types of cropland areas decreased from 2000 to 2009 whereas forest and grassland increased in this period. Actually, the increase in forest and grassland was mainly due to a series of national ecological conservation policies, such as the Grain for Green Project, which started in 1999.

Changes in biodiversity

Habitat quality is used to indicate how well a grid cell can support wildlife and natural vegetation over time (Himlal et al., 2014). In this study, we defined raster threats of the watershed as cropland, rural residential areas, urban, rotation forestry, and roads (including primary roads, secondary roads, and light roads). These variables were extracted from land use/cover maps and represent anthropogenic drivers of land-use change on the watershed landscape (Figure 3). The spatial mapping methods for raster threats of 2000, SP1, and SP2 were similar with methods for threat mapping in 2009.
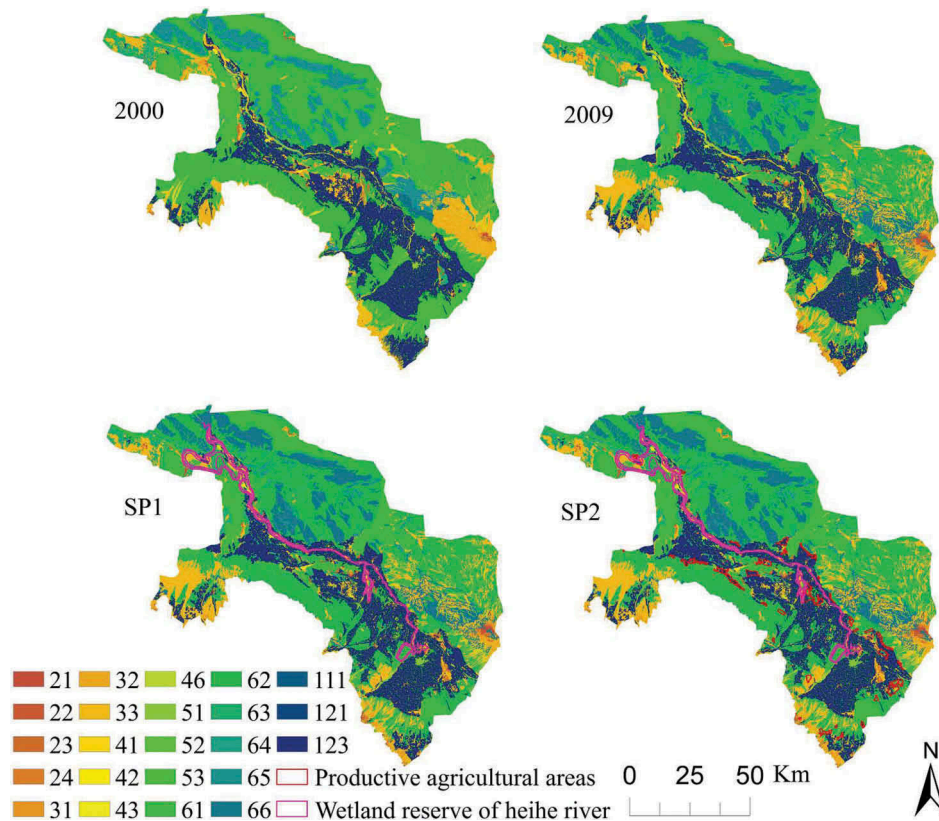
Figure 2. LUCC maps of 2000, 2009, SP1, and SP2 in the artificial desert watershed (modified from Liang and Liu, 2014).

Table 3. Area and proportion of LUCC in the artificial desert watershed from 2000 to 2009.

LUCC	2000 Areas (km^2)	2000 Areas (%)	2009 Areas (km^2)	2009 Areas (%)
Forest	11.81	0.10	30.95	0.27
Spinney	33.76	0.30	41.09	0.36
Open woodland	37.91	0.34	71.90	0.64
Other woodland	1.21	0.01	12.58	0.11
High-coverage grassland	21.17	0.19	278.93	2.47
Middle-coverage grassland	207.56	1.84	355.44	3.15
Low-coverage grassland	933.77	8.27	1648.36	14.59
Canal	125.58	1.11	52.79	0.47
Lake	1.45	0.01	2.57	0.02
Glaciers and permanent snow	31.11	0.28	18.56	0.16
Shallow	186.86	1.65	143.87	1.27
Urban areas	13.39	0.12	16.52	0.15
Rural areas	115.10	1.02	124.43	1.10
Other construction areas	14.02	0.12	16.87	0.15
Desert	1663.95	14.73	1293.61	11.45
Gobi	3988.62	35.31	3621.81	32.05
Saline-alkali fields	123.70	1.10	685.68	6.07
Marsh	239.11	2.12	17.05	0.15
Bare areas	46.42	0.41	160.46	1.42
Rock	1225.66	10.85	1011.79	8.95
Paddy field	0.01	0.00	0.04	0.00
Mountain cropland	22.89	0.20	1.57	0.01
Plains cropland	2250.71	19.92	1688.9	14.99
Total	11,295.77	100	11,295.77	100

Based on different raster maps and other input data, biodiversity change was first analyzed at the pixel scale. Biodiversity followed a similar distribution pattern as land use/cover in the watershed (Figures 2 and 4); habitat quality values were mainly located in the cropland, forest, and grassland land-cover types, which occupied over 31.17% of the watershed in 2000 and 41.71% in 2009 watershed. These areas are critical for sustained ecosystem service provisioning and biodiversity conservation. As shown in Figure 4, there was a large amount of spatial variability in the distribution of watershed biodiversity change.

Biodiversity was very dynamic in the southeast to central watershed and decreased moderately in the south, an area that had an expansion of cropland from 2000 to 2009. Scenario SP2 resulted in a greater increase in biodiversity compared to SP1. These results show that local managers should consider improvement of biodiversity where the
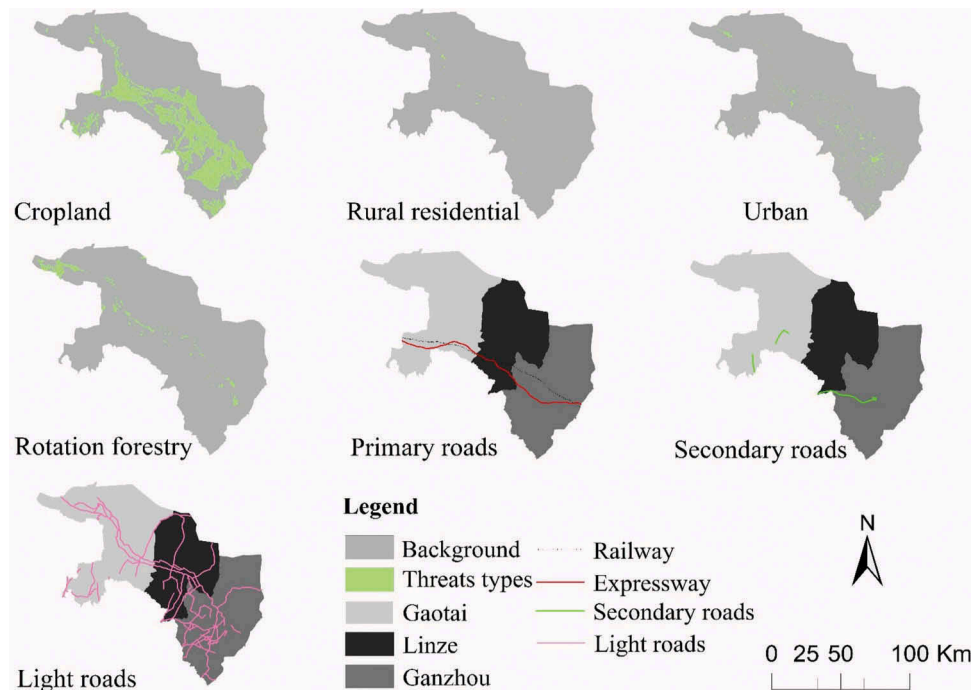
Figure 3. Spatial distribution of different threats based on the LUCC map of 2009 in the artificial desert watershed.
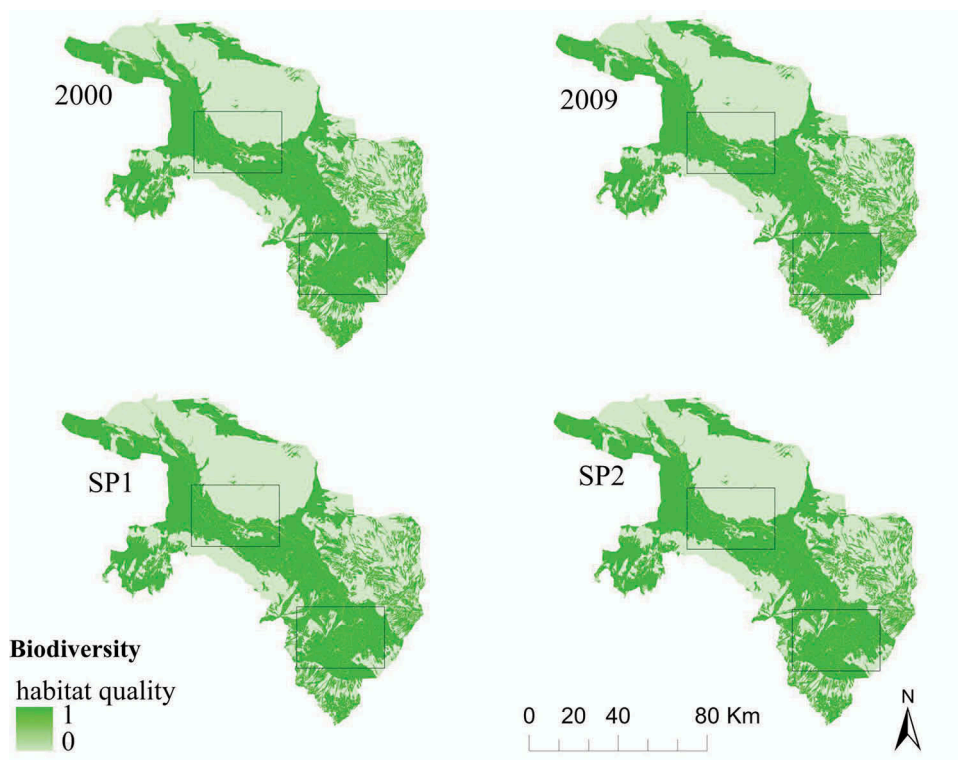

Figure 4. Spatial distribution of biodiversity in 2000, 2009, SP1, and SP2 at the pixel scale.

current status is too low and work to maintain the current ecosystem status in areas with high levels of natural capital. We emphasize here the importance of spatially explicit results at the pixel scale, representing spatial heterogeneity in the quantity and quality of biodiversity and its effect on ecosystem service provisioning is critical to decision-making (De Groot et al., 2010).

Biodiversity change of the artificial desert watershed was also analyzed at the regional scale. Biodiversity habitat quality scores were calculated by summing the value of each cell across the landscape. Regionally, biodiversity experienced a slight decrease in total value of habitat quality for both the 2000–2009 period and under the management scenarios. Biodiversity experienced less of a decrease

from 2000 to 2009 (total decrease of habitat quality value was 2.76) compared to SP1 (total decrease of habitat quality value was 99.81) and SP2 (total decrease of habitat quality value was 83.02).

LUCC impact on biodiversity

A qualitative assessment of Land Use and Coverage Change (LUCC) processes and their impact on biodiversity and ecosystem services (Figure 5(a)) indicated that the study area was covered with seven typical land use types (NP – nature protection, RC – road construction, FC – farmland construction, MI – mining integration, UR – urbanization, WD – wetland development, FI – fruit and vegetable industry; Figure 5(b)) that supported biodiversity and supplied a wide range of ecosystem services (Liang et al., 2013). After rapid growth of the population and economy in the 1970s, the majority of the landscape was cleared, resulting in increased agriculture production at the expense of other ecosystem services. After 2010, the reconfigured landscape included a combination of agriculture and nature protection. The typical land-use patterns after 2010 indicate an overall positive impact on a number of ecosystem services and the potential for integrating environmental planning, agro-forestry and extensive agriculture production, and urbanization (Figure 5(b)).

Actually, biodiversity was dynamic due to varying levels of change at different scales. Some regions showed improvements and others decreases in biodiversity. In fact, at the scale of the entire watershed, an obvious drop in ecosystem function is not immediately apparent; the change trend shows a decline in biodiversity in the rapidly developing areas. According to local government planning, a rapid population increase and urbanization are expected to occur over the next few decades (ZSB 2009); so, stakeholders in the watershed could experience the effects of land-use management change on biodiversity, particularly in terms of agricultural expansion and related ecological conservation approaches, such as in SP1 and SP2.

Discussion

In this study, we found that the artificial desert watershed is likely to experience ecosystem biodiversity degradation or improvement due to land-use
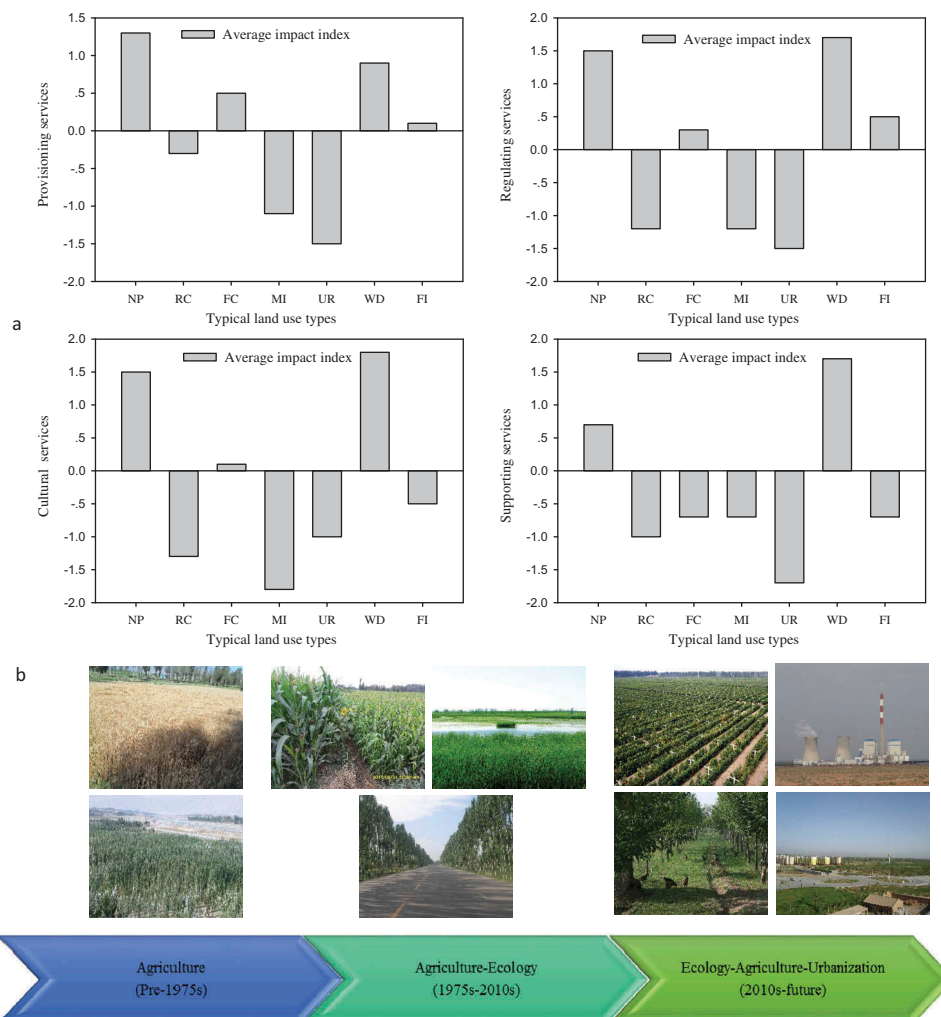
Figure 5. A qualitative assessment of LUCC impact on ecosystem services (figure data provided by Liang et al., 2013) and a typical land-use transition in the artificial desert watershed among multiple ecosystem services: pre-1975, 1975–2010s, and 2010–future landscape.

management changes based on the policies that have been implemented. As shown by maps of the relations between land-use changes and the spatial distribution of biodiversity (Figures 2 and 4), conservation should be given priority in areas surrounding the Grain for Green Project in the central and southeast regions.

One would predict that the optimum effect on biodiversity on this landscape would result from natural resource management watershed. However, the simulated changes in biodiversity under SP1 and SP2 appear to result from regional-scale phenomena rather than from local-level land-use changes, although more process-based modeling of trade-off effects on biodiversity and ecosystem services is necessary to confirm this conclusion. In the artificial desert watershed, the decrease of habitat quality from 2009 to SP1 is especially relevant to the National Wetland Reserve of Heihe River. Thus, land-use management policies should be implemented based on more critical monitoring and prediction of the drivers of habitat change in this watershed.

As pointed out above, previous local studies have analyzed the LUCC effects of different land management scenarios (e.g., Liang and Liu, 2014). Using LUCC classifications based on different landscapes posed a challenge in this study. The categories of forest, grassland, wetland, and cropland are important sources of interaction with biodiversity and ecosystem services provisioning in the artificial desert watershed (Figure 5(a)). Simulating the scale effect of input data for biodiversity change is a key issue given the diverse land-use types in the artificial desert watershed.

We focused on the change of biodiversity in our analysis. However, the influence of human activities (e.g., RC, UR, and WD) is a key element with great effects on biodiversity watershed. We must consider the relationship between the selected indicators and the local human activities of this watershed. For instance, habitat quality is closely related to the numbers of tourists in the area. In 2011, more than 3.5 million tourists came to visit (data source: Bureau of Tourism in Zhangye government, http://tour.zhangye.gov.cn/index.aspx). The tourists mainly want to visit mature ecosystems and are especially interested in hiking and watching wildlife within and near the National Wetland Reserve of the Heihe River watershed. As shows in this study, the habitat quality indicator for biodiversity varies according to location and social context. The watershed is undergoing a transition from rapid economic development to sustainable development and use of natural resources. Generally, the public strongly wishes to prevent the extinction of endangered species in the fragile watershed desert

ecosystem. Thus, the use of spatial analysis to illustrate explicit changes in ecosystems is useful in fostering biodiversity awareness and the need for decision-making at different scales.

Conclusions

This study combined different LUCC with biodiversity assessment models in the GIS-based tool InVEST to determine the patterns of land-use change from 2000 to 2009 and under two potential land-use management scenarios. We also examined the effects on habitat quality at the pixel level and at regional scales in the Gaotai, Linze, and Ganzhou watershed. The empirical land-use spatial mapping approach of different scenarios was based on analysis of the key management strategies used by local government in addressing the drivers of past and future land-use changes. Thus, this integrated approach allowed us to predict possible future biodiversity change in a spatially explicit way for the benefit of local communities.

From 2000 to 2009, habitat quality was predicted to decrease slightly in the central and southeast regions at the pixel scale. Thus, with no conservation policies and land management in place, specific areas of the watershed were expected to experience a decline in biodiversity, especially in areas dominated by agriculture. In the artificial desert watershed, local management practices often play the most important role in maintaining biodiversity and specific ecosystem functions. Although the land-use conservation scenario SP2 was predicted to improve biodiversity indicators compared to SP1, the wetland conservation scenario resulted in slight decreases in biodiversity. Thus, our findings indicate that land-use decisions for the wetland conservation in watershed are particularly important for future management of the integrated biodiversity and ecosystem services of the entire watershed.

Based on the model simulation results, conservation measures, which are key features in the management of the wetland reserves and the Grain for Green Project landscape, are recommended for the provisioning of biodiversity. As observed in this study, land-use management strategies under different scenarios can affect biodiversity of oasis both at local and regional scales. Our findings illustrate the spatial and temporal change of biodiversity that occurs within a typical artificial desert watershed as a result of land-use management strategies related to biodiversity conservation. These simulation results will support land-use decision-making and planning of local governments by highlighting the potential trade-offs and outcomes in the watershed.

Disclosure statement

No potential conflict of interest was reported by the authors.

References

Baral, H., J. K. Rodney, K. S. Sunil, E. S. Nigel, and K. Sabine. 2014. "Spatial Assessment and Mapping of Biodiversity and Conservation Priorities in a Heavily Modified and Fragmented Production Landscape in North-Central Victoria, Australia." *Ecological Indicators* 36 (1): 552–562. doi:10.1016/j.ecolind.2013.09.022.

Brooks, T. M., R. A. Mittermeier, D. F. Gab, J. Gerlach, M. Hoffmann, J. F. Lamoreux, C. G. Mittermeier, J. D. Pilgrim, and A. S. L. Rodrigues. 2006. "Global Biodiversity Conservation Priorities." *Science* 313: 58–61. doi:10.1126/science.1127609.

Butchart, S. H., M. Walpole, B. Collen, A. Van Strien, J. P. Scharlemann, R. E. Almond, J. E. Baillie, et al. 2010. "Global Biodiversity: Indicators of Recent Declines." *Science* 328 (5982): 1164–1168. doi:10.1126/science.1187512.

De Groot, R. S., R. Alkemade, L. Braat, L. Hein, and L. Willemen. 2010. "Challenges in Integrating the Concept of Ecosystem Services and Values in Landscape Planning, Management and Decision Making." *Ecological Complexity* 7: 260–272. doi:10.1016/j.ecocom.2009.10.006.

Eigenbrod, F., B. J. Anderson, P. R. Armsworth, A. Heinemeyer, S. F. Jackson, M. Parnell, C. D. Thomas, and K. J. Gaston. 2009. "Ecosystem Service Benefits of Contrasting Conservation Strategies in a Human-Dominated Region." *Proceedings of the Royal Society B: Biological Sciences* 276 (1669): 2903–2911. doi:10.1098/rspb.2009.0528.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, et al. 2005. "Global Consequences of Land Use." *Science* 309: 570–574. doi:10.1126/science.1111772.

Geneletti, D. 2013. "Assessing the Impact of Alternative Land-Use Zoning Policies on Future Ecosystem Services." *Environmental Impact Assessment Review* 40: 25–35. doi:10.1016/j.eiar.2012.12.003.

Goldstein, J. H., G. Caldarone, T. K. Duarte, D. Ennaanay, N. Hannahs, G. Mendoza, S. Polasky, S. Wolny, and G. C. Daily. 2012. "Integrating Ecosystem-Service Tradeoffs into Land-Use Decisions." *Proceedings of the National Academy of Sciences* 109: 7565–7570. doi:10.1073/pnas.1201040109.

Guan, Q. F., and K. C. Clarke. 2010. "A General-Purpose Parallel Raster Processing Programming Library Test Application Using a Geographic Cellular Automata Model." *International Journal of Geographical Information Science* 24 (5): 695–722. doi:10.1080/13658810902984228.

Himlal, B., J. K. Rodney, K. S. Sunil, E. S. Nigel, and K. Sabine. 2014. "Spatial Assessment and Mapping of Biodiversity and Conservation Priorities in a Heavily Modified and Fragmented Production Landscape in North-Central Victoria, Australia." *Ecological Indicators* 36: 552–562. doi:10.1016/j.ecolind.2013.09.022.

Kandziora, M., B. Burkhard, and F. Müller. 2013. "Interactions of Ecosystem Properties, Ecosystem Integrity and Ecosystem Service Indicators—A Theoretical Matrix Exercise." *Ecological Indicators* 28: 54–78. doi:10.1016/j.ecolind.2012.09.006.

Liang, Y., L. Liu, and J. Huang. 2017. "Integrating the SD-CLUE-S and Invest Models into Assessment of Oasis Carbon Storage in Northwestern China." *PLoS One* 12 (2): e0172494. doi:10.1371/journal.pone.0172494.

Liang, Y. J., and L. J. Liu. 2014. "Modeling Urban Growth in the Middle Basin of the Heihe River, Northwest China." *Landscape Ecology* 29 (10): 1725–1739. doi:10.1007/s10980-014-0089-9.

Liang, Y. J., Z. M. Xu, F. L. Zhong, and X. Y. Song. 2013. "An Spatial Ecosystem Services Approach Based on LUCC: A Case Study of Ganzhou District of Zhangye City." *Acta Ecologica Sinica* 33 (15): 4758–4766. doi:10.5846/stxb201205140708.

Minin, E. D., A. Soutullo, L. Bartesaghi, M. Rios, M. N. Szephegyi, and A. Moilanen. 2017. "Integrating Biodiversity, Ecosystem Services and Socio-Economic Data to Identify Priority Areas and Landowners for Conservation Actions at the National Scale." *Biological Conservation* 206: 56–64. doi:10.1016/j.biocon.2016.11.037.

Nelson, E., H. Sander, P. Hawthorne, M. Conte, D. Ennaanay, S. Wolny, S. Manson, and S. Polasky. 2010. "Projecting Global Land-Use Change and Its Effect on Ecosystem Service Provision and Biodiversity with Simple Models." *PLoS One* 5 (12): e14327. doi:10.1371/journal.pone.0014327.

Polasky, S., E. Nelson, D. Pennington, and K. Johnson. 2011. "The Impact of Land-Use Change on Ecosystem Services, Biodiversity and Returns to Landowners: A Case Study in the State of Minnesota." *Environmental and Resource Economics* 48: 219–242. doi:10.1007/s10640-010-9407-0.

Rands, M. R. W., W. M. Adams, L. Bennun, S. H. M. Butchart, A. Clements, D. Coomes, A. Entwistle, et al. 2010. "Biodiversity Conservation: Challenges Beyond 2010." *Science* 329: 1298–1303. doi:10.1126/science.1189138.

Steffen, W., A. A. Burbidge, L. Hughes, R. Kitching, D. Lindenmayer, W. Musgrave, M. S. Smith, and P. A. Werner. 2009. *Australia's Biodiversity and Climate Change*. Melbourne: CSIRO Publishing.

Tallis, H. T., T. Ricketts, A. D. Guerry, S. A. Wood, R. Sharp, E. Nelson, D. Ennaanay, et al. 2013. *Invest 2.6.0 User's Guide*. Stanford: Natural Capital Project.

Wilson, K. A., E. Meijaard, S. Drummond, H. S. Grantham, L. Boitani, G. Catullo, L. Christie, et al. 2010. "Conserving Biodiversity in Production Landscapes." *Ecology Applications* 20(6): 1721–1732.

Xiao, Y., G. D. Xie, C. X. Lu, X. Z. Ding, and Y. Lu. 2005. "The Value of Gas Exchange as a Service by Rice Paddies in Suburban Shanghai, PR China." *Agriculture, Ecosystems & Environment* 109: 273–283. doi:10.1016/j.agee.2005.03.016.

Xie, G. D., Y. Xiao, L. Zhen, and C. X. Lu. 2005. "Study on Ecosystem Services Value of Food Production in China." *Chinese Journal of Eco-Agriculture* 13 (3): 10–13. (In Chinese).

Xie, G. D., L. Zhen, C. X. Lu, Y. Xiao, and W. H. Li. 2010. "Applying Value Transfer Method for Eco-Service Valuation in China." *Journal of Resources and Ecology* 1 (1): 51–59.

Xuan, G., C. C. Nicholas, T. Piotr, E. N. Scott, W. B. Christopher, and S. John. 2017. "Regional Mapping of Vegetation Structure for Biodiversity Monitoring Using Airborne Lidar Data." *Ecological Informatics* 38: 50–61. doi:10.1016/j.ecoinf.2017.01.005.

Zhao, W. Z., B. Liu, and Z. H. Zhang. 2010. "Water Requirements of Maize in the Middle Heihe River Basin, China." *Agricultural Water Management* 97 (2): 215–223. doi:10.1016/j.agwat.2009.09.011.

ZSB (Zhangye Statistic Bureau). 2000. *Yearbook of National Economy and Social Development of Zhangye City*, 34–38. Zhangye City: Zhangye Statistic Bureau. (In Chinese).

ZSB (Zhangye Statistic Bureau). 2009. *Yearbook of National Economy and Social Development of Zhangye City*, 43–49. Zhangye City: Zhangye Statistic Bureau. (In Chinese).