

Modeling of ecosystem services informs spatial planning in lands adjacent to the Sarvelat and Javaherdasht protected area in northern Iran



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ABSTRACT

Dynamic models of ecosystem services supply and scenario analysis of changes in multiple services are being increasingly used to support land use planning and decision making. This approach reduces potential and real conflicts among various stakeholders potentially creating win–win solutions for all. It is particularly applicable in areas where insufficient land for agriculture and settlements is resulting in high rates of conversion of natural forest and grasslands. We quantified and mapped multiple ecosystem services, including habitat provision as a proxy for biodiversity, carbon storage and sequestration, and water balance and supply in the Sarvelat and Javaherdasht region of the globally-significant Hyrcanian (Caspian) forests in northern Iran using the Integrated Valuation of Ecosystem Services and Tradeoffs tool. This region is experiencing a rapidly increasing rate of forest conversion and as a result, the protected area located within the study landscape is threatened by human encroachment. Plausible future landscapes were modeled under three scenarios: (i) business as usual; (ii) protection-based zoning which reflects an expansion of the protected area boundary to prevent land use changes; and (iii) collaborative zoning through redefining the protection boundary simultaneously with an adjustment to meet local stakeholders' objective of expansion of anthropogenic cover. The results showed that the collaborative zoning scenario would best contribute to effective policy because it presents a more rational spatial configuration of the landscape maintaining the provision of ecosystem services. This scenario may lead to reduced environmental impacts while achieving less conflict between the government and local communities. These results will help to inform and shape natural resource management policies in Iran and is applicable elsewhere in the world.

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1. Introduction

Ecosystems and their associated biological diversity provide a wide range of ecosystem services (ES) which benefit human

well-being (Bastian et al., 2013; MEA, 2005). Internationally, the importance of maintaining ES through the conservation of biodiversity is recognized through Aichi Target 11, which falls under Goal C of the Convention on Biological Diversity (CBD) Strategic Plan for 2011–2020. This commits all parties to conserve areas of particular importance for biodiversity through well-connected systems of protected areas to enhance the benefits to communities from their ES (Woodley et al., 2012). Biodiversity underpins the continuous provision of ES which is essential for human survival and well-being (Balvanera et al., 2012) and the success of the economy

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and social welfare. The Millennium Ecosystem Assessment (MEA, 2005), classifies ES as *provisioning* (e.g. food production), *regulating* (e.g. carbon sequestration), *cultural* (e.g. recreation) services that directly affect people, and *supporting* services (e.g. provisioning of habitat and primary production) which are vital for maintaining the other services.

Despite their obvious importance for quality of life, ES are declining in many ecosystems due to human encroachment (MEA, 2005). While population growth that increased the demand for ES, the capacity to supply such services is decreasing due to extensive human alteration of natural ecosystems in recent decades (DeFries et al., 2004; Foley et al., 2011). There are numerous factors which affect the supply-demand interaction of ES within a complicated social-ecological system (Nassi and Löffler, 2015) and hence, there is an increasing focus on spatial modeling of ES and drivers of change to help develop land-use policies and support more effective decision-making (Bélair et al., 2010; Paudyal et al., 2016; Power, 2010).

Human interventions which result in the loss of ES occur mainly through changing land use/land cover (Bennett, 2005; Bennett and Balvanera, 2007; Bhatta et al., 2015; Zarandian et al., 2016) here abbreviated to land cover. For example, management decisions that are taken to increase food production through agricultural development, or expansion of settlements through forest clearing, may negatively impact ES such as climate regulation and habitat availability (Baral et al., 2014a). Therefore, after assessing the current state of ES supply in a given area, prediction of the impacts of changes in ES supply, based on future plausible scenarios, can help decision-making and reduce risks (Raudsepp-Hearne et al., 2010; Swetnam et al., 2011; Wollenberg et al., 2000).

With the availability of new predictive tools, it is possible to predict the impact of changes in land use policies and different management regimes and decisions on ES supply. ES are incorporated in land use planning and conservation investment through robust quantification of the stocks and flows of such services and placing economic values on them (Kareiva et al., 2007; Palomo et al., 2013; Tardieu et al., 2013; Wittmer and Gundimeda, 2012). At a broader environmental policy level, planners need to understand where, when, and what ES are supplied by a specific landscape so that the effects of policy options and decisions can be monitored (Baral et al., 2014a; Bhatta et al., 2016; Crossman et al., 2013; García-Nieto et al., 2013; Mubareka et al., 2013). One of the major challenges is to develop appropriate methods for ES mapping and understand how regulating and supporting services are affected by human activities in the landscape as there are inconsistencies between existing ES quantification and mapping methods, hindering their application in planning and management processes (Baral and Holmgren, 2015; Crossman et al., 2013; Paudyal et al., 2016). In addressing these challenges, standard dynamic ES models have been developed to facilitate mapping of ES supply, providing powerful tools for processing spatially and temporally complex data (Burkhard et al., 2012a,b; Crossman et al., 2012; Martínez-Harms and Balvanera, 2012) and analysis of tradeoffs between multiple ES under different scenarios (Boumans et al., 2002; Nelson et al., 2009). Scenario modeling and analysis facilitates assessments in complex systems such as ecosystems and has become an important part of integrated environmental assessments, including those which have been carried out by the Intergovernmental science-policy platform on biodiversity and ecosystem services (IPBES) and the Millennium Ecosystem Assessment (Robinson et al., 2016).

Many studies have been carried out on the effects of ecosystem conversion (e.g. forests, wetlands and rangelands) on ES supply such as the consequences of land use change for carbon sinks (Eaton and Lawrence, 2009), primary production (Portela and Rademacher, 2001) and water flows and quality (Uriarte et al., 2011). However, in most developing countries, such as Iran,

environmental studies using an ES approach are in their infancy, and as yet there has been no study conducted with an explicit focus on multiple ES modeling. This research is the first practical step in the application of existing models in a quantitative ES assessment in the Iranian Sarvelat and Javaherdasht forested landscape, including its protected area, where native forest land conversion for housing and farm development has accelerated in the last three decades. Elsewhere, we have assessed the status and trends of key ES in this region using a rapid, qualitative and participatory approach, including interviews with local households and experts in combination with an assessment of remote sensing data on land cover to identify and map priority ES in GIS (Zarandian et al., 2016). The results showed that, although food production and recreation have greatly increased in recent decades, other services, in particular, timber production, habitat provision, and water purification and supply, gradually are being lost (Zarandian et al., 2016). Since the current process of land conversion has resulted in increased conflict between different stakeholders, including conservation organizations and landowners, here in this current study, we present a possible win-win solution to reduce conflicts and seek a compromise between the land cover changes and nature conservation based on an ES approach.

Our specific aim was to predict plausible future land use scenarios through mapping the spatial distribution and quantification of habitat provision, carbon sequestration and water balance ES in the current landscape. Through the development of these future land use scenarios we aimed to inform policy makers on how they may best formulate effective land use policy while resolving potential stakeholder conflicts.

2. Materials and methods

2.1. Study area

The Sarvelat and Javaherdasht forested landscape covers an area of 55,840 ha and is located between 36°48'–37°03' N latitude and 50°22'–50°45' E longitude, lying within the boundaries of two provinces, Gilan and Mazandaran, in the northern part of Iran (Fig. 1). Iran, declared 21,254 ha of this area as a protected area in 1999, as this area plays an important ecological role in the provision of ES such as water balance and climate regulation (Zarandian et al., 2016).

This region is mountainous with an altitudinal range of zero to 3550 m above sea level. The lowlands have a humid temperate climate with an average annual temperature of 14°C and average annual precipitation of 1150 mm. The Hyrcanian (Caspian) mixed forest of northern Iran and southern Azerbaijan is a deciduous broad-leaved forest classified as ecoregion number 78 in WWF's Global 200 Ecoregions (Olson and Dinerstein, 1998). The area contains remnants from the Tertiary period and has around 150 endemic species of trees and shrubs, and 60 mammal, 340 bird, 67 fish, 29 reptiles, and 9 amphibian species. The landscape lies along an important bird migratory route between Russia and Africa and is listed as an important bird area (IBA) (Caspian Hyrcanian Forests Project, 2015). It has been listed by Iran as a future site for nomination as a World Heritage site (UNESCO, 2015). About 86% is covered with forest most of which is located at an altitude of 400–2000 m. Other land uses are orchards/gardens (mainly citrus groves), farmlands (rice) and human settlements which, respectively, account for approximately 1%, 4% and 3% of all land cover and are located in the lowlands. Approximately 4% of the landscape is bare rock, mostly located at altitudes above 2,800 m.

Since the establishment of a protected area in the southern part of the landscape, and given that this area is bordered on the north by the Caspian Sea, there is limited flat land available for

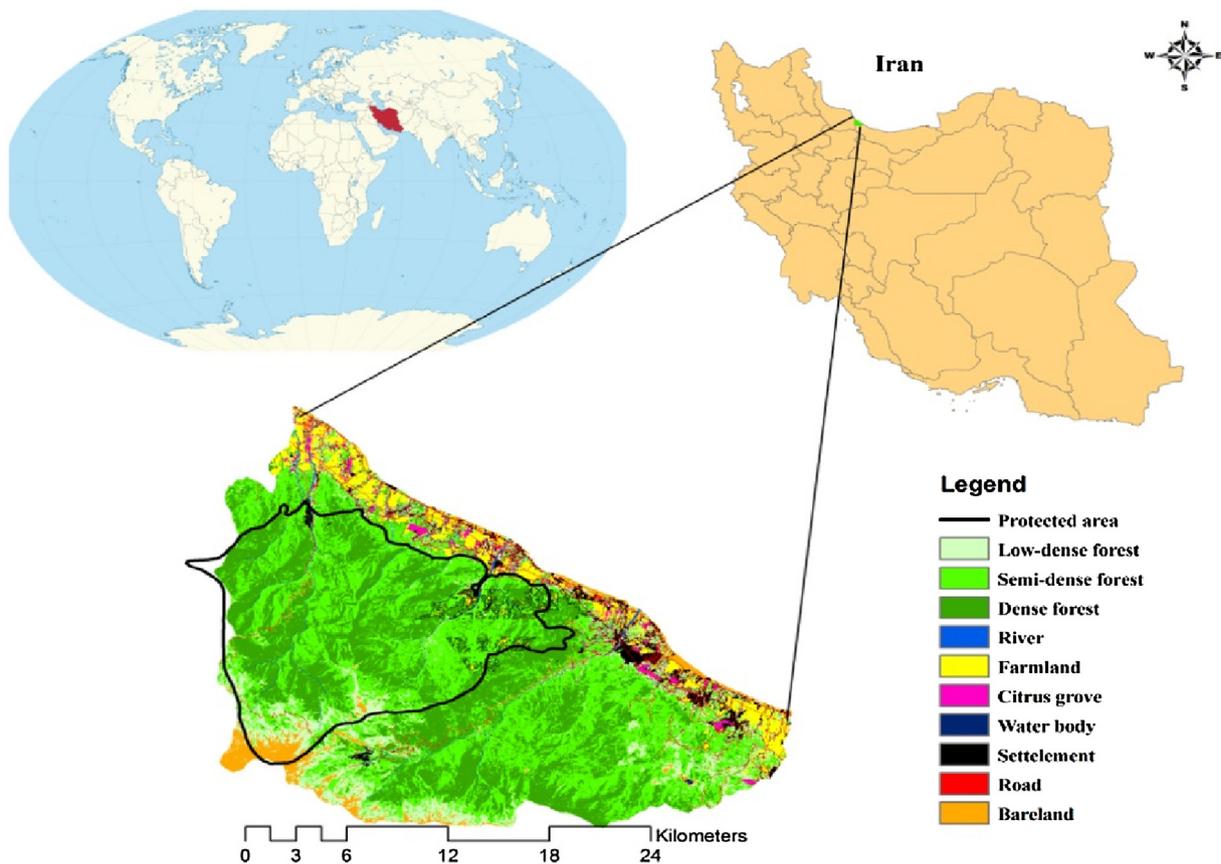


Fig. 1. Location of the Sarvelat and Javaherdasht landscapes, the boundaries of the protected area, and location of the major land cover types (Zarandian et al., 2016).

new agricultural and urban development. As a result, the study area is highly vulnerable to land cover changes with forest conversion accelerating in the uplands and near the border of the protected area in recent decades (Zarandian et al., 2016). These changes have resulted in increased conflicts among the various stakeholders including local residents, conservation organizations, and construction companies. In the eastern half of land adjacent to the protected area there are native, but non-protected forest lands, which are at risk of degradation and clearing if the existing expansion trend continues.

2.2. Methods

Transitional participatory scenario building was used to identify the concerns of local residents in changes in land cover and was carried out through direct interviews of heads of households with the households selected randomly. To determine the appropriate sample size of households for interview we used the Raosoft sample size calculator (<http://www.raosoft.com/samplesize.html>). To apply this tool, the user predetermines two factors: margin of error (or confidence intervals) and confidence level. The margin of error is the deviation you allow between the opinions of your respondents and the opinion of the entire population. Reducing the margin of error requires increasing the sample size, which needs more resources such as money and time. The population within our study area was relatively homogenous because the landscape was a small pilot area within a larger geographical region with common socio-cultural characteristics. Moreover, most of the local population have similar livelihoods: farming or providing different services for tourists. Accordingly, we allowed an error margin of

10% because it was not necessary to choose the maximum accuracy rate for this factor which is 5%. Also, we used the 95% confidence level, which is used by most researchers, and hence selected 95 households from the total of 6762 for interview.

We also used tools from the suite of open-source software models in Integrated Valuation of Ecosystem Services and Tradeoffs package (InVEST) (<http://www.naturalcapitalproject.org/InVEST.html>) to assess spatial-temporal changes of multiple ES supply associated with land cover changes (Sharp, 2014). This includes a set of ecosystem models for terrestrial, marine, and freshwater that uses production equations to estimate changes in biodiversity and ES. Correlations between them are carried out under various human populations, land use and climate change scenarios (Arkema et al., 2013; Bai et al., 2011; Guerry et al., 2012; Kareiva et al., 2011; Nelson et al., 2009; Polasky et al., 2011; Tallis and Polasky, 2009). Various tools from the package have been used in different parts of the world to map multiple or single ES (see Baral et al., 2013; Müller and Burkhard, 2012; van Oort et al., 2015; Zarandian et al., 2016). For example, in Central Sumatra, Indonesia, InVEST was used to map and quantify the current provision of ES under two alternative scenarios (Bhagabati et al., 2012). It has also been used to assess tradeoffs among biodiversity and other ES in Australia (Baral et al., 2014a). By using InVEST models, marine planners in western Canada were able to use science to help resolve conflicts among different interest groups and make implicit decisions more explicit (Guerry et al., 2012).

Using the production function approach in these models can demonstrate how these structural changes cause changes in the functioning of ecological processes or the provision of ES (Ruckelshaus et al., 2013).

This part of the study includes four key steps:

- (i) Mapping of the current land cover using satellite imagery data.
- (ii) Prediction of land cover and protection boundary transitions with the engagement of related stakeholders and mapping it under different scenarios using the InVEST Scenario Generator supporting tool.
- (iii) Mapping and quantification of multiple ES under current and plausible future landscapes using the InVEST Biodiversity (Habitat Quality), Carbon Storage and Sequestration and Water yield models.
- (iv) Comparison of model outputs to suggest the most effective policy associated with landscape land cover changes.

Although the economic valuation of water and carbon services was possible at the final stage of this study based on the resulting quantified data, we did not value the ES assessed, because there were no financial mechanisms for the potential water surplus for agricultural and domestic uses as well as no market for carbon. Instead, we provided biophysical and economic information that could be used in a valuation study once beneficiaries, price signals, and replacement costs are identified.

2.2.1. Plausible future scenarios

Three plausible scenarios were defined and extrapolated to the year 2025:

2.2.1.1. Business as usual scenario (BAU). This scenario was based on the results of our previous study (Zarandian et al., 2016) of past land cover transitions as well as identification of the present needs of local residents for land conversion. It assumes continuation of the existing trend of conversion of natural forest cover to agricultural land and settlements. Comparison of recent (here we call this 'current') (2013) and baseline (2003) land cover maps showed that dense forest cover is gradually being replaced by low-density and semi-dense forest covers, we suggest, reflecting increasing levels of forest disturbance. In this decade low-density and semi-dense forest cover have increased by 80% and 17%, while dense forest has been reduced by 26%. Although not all the changes were due to land use change with other factors such as poor forest management and illegal timber logging being involved, the trend generally reflects a rapid decline of natural forest cover.

The results of our household survey (Zarandian et al., 2016) indicated that the dramatic growth in land and housing prices and the increasing demand from local people to build settlements and develop farmlands in order to generate more revenue have been the key drivers of land cover changes in the past decade. The lack of efficient land use planning in the region suggests that the BAU scenario is highly plausible. Because of the limited availability of land downstream, local residents will seek to develop currently forested areas outside the protected areas and in some eastern uplands. Although legal protection of the forested area is not very effective, we chose to assume that the forest cover inside the protected area will remain unchanged. Accordingly, the incremental increases for four land cover classes were assumed to be: 60% for farm fields; 70% for citrus groves; 90% for human settlement; and 5% for road networks.

2.2.1.2. Protection-based zoning scenario (PBZ). This scenario reflects the interest of government conservation-focused organizations in response to the existing trend of land conversion with the aim of preventing further encroachment in upland forests. To develop this scenario we held a consultative meeting with representatives from both the Department of Environment (DOE) and the Forests, Rangelands, and Watershed Organization (FRWO) which are the main conservation bodies in Iran. The BAU scenario was presented and they were asked to how their departments

might react. As their environmental policies are mostly regulatory with a command and control approach, most representatives suggested an expansion of protected boundaries to prevent habitat loss.

This means that some of the surrounding unprotected natural forest covers would be incorporated into the current protected area. In this scenario, the protected areas would increase from the current 21,254 ha to 37,554 ha in the future. Due to high demand for land conversion, it is anticipated that more conflict between the two groups will result while at the same time exacerbating the use of land downstream close to the border of the protected area.

2.2.1.3. Collaborative zoning scenario (CZ). It is unusual in Iran to plan and make land use decisions based on a participatory approach with compromises among both governmental and non-governmental stakeholders. In addition, unlike many developing countries, non-governmental conservation bodies are in their infancy in Iran and have little involvement and impact at the local level. So this scenario was defined hypothetically in a situation where the needs of local people were also considered. Both side's interests were adjusted to reach to an optimal, acceptable and applicable policy. In fact, this scenario aims to show to policy makers a third solution can be achieved by collaboration among all stakeholders balancing the objectives of both government and local residents and represents a win-win policy. In this scenario, the boundary of the protected area is redefined through consultation with local stakeholders. Accordingly, 7314 ha of the current protected area downstream, which is vulnerable in terms of habitat condition because of its proximity to current agriculture and urban settlements would be excluded from protection and made available for future planned development. The incremental changes in four land cover classes were assumed to be: 50% for farm fields; 66% for citrus groves; 50% for human settlement; and 5% for road networks. In exchange 15,157 ha of the current unprotected but high-quality forest cover in the eastern part of the landscape (more than twice the area of the excluded protected area), which are at risk of degradation in the future, would be added to the protected area. Under this scenario, while the size of protected areas will increase, restrictions on land availability are largely eliminated in response to the needs of local stakeholders. Therefore, it will prevent more conversion of forest cover in upland areas which are sources of ES and it reduces threats to ES supply in the future. Table 1 shows land cover transitions and their quantities of changes under each plausible future scenario.

2.2.2. Models and tools

2.2.2.1. InVEST scenario generator model. To visualize plausible future land cover under each of the three scenarios, InVEST Scenario Generator Model was used (Sharp, 2014). This tool analyses land suitability based on input data. These data include the quantity of land cover changes, transition likelihoods, land cover priorities or weights from the perspective of stakeholders, and physical factors that determine land suitability such as slope, elevation, soil type, distance to roads and markets and rainfall distribution. The likelihood that given parcels of land cover classes will be converted from one land cover type to another (transition likelihood) was defined based on the recent trend of changes and was ranked on a scale of 0–1 (0 means unlikely and 1 is the most likely). Since in some cases, multiple land cover objectives compete for a single land parcel (or pixels in a raster grid map), the land cover types were ranked according to their priorities. The model compares different classes of land cover using a pair-wise comparison matrix with a nine-point continuous scale (Saaty, 1977) and these are ranked based on an analytic hierarchy process. Then physical and environmental factors were weighted on a scale of 0 (unsuitable) to 1 (extremely suitable). These include elevation as the determinant factor in the

Table 1
Percentages of land cover changes and the area covered of natural/human-made assets under each future plausible scenario.

Scenarios	Areas of different land covers (ha)			
	Natural assets (Different density of forests, rivers, water bodies, and bare lands)	Agricultural assets (Farmlands, citrus groves)	Human-made infrastructures (Road networks, settlements)	Conservation (Lands under protected area)
BAU	46,096.47	3478.68	3985.74	21,254
PBZ	49,669.11	2967.39	3192.66	37,554
CZ	49,050.54	3301.11	3477.51	29,097

development of human settlements with a weight of 0.8. Elevation and distance to roads were the determining factors in relation to farmland with weights of 0.9 and 0.2, respectively, as well as distance to roads for citrus groves with a weight of 0.7. To incorporate the effect of the proximity factor in the conversion of land cover types, we considered the maximum effective proximity distance for forest, farmland and citrus groves, settlements, and roads to be equal, at 10,000, 3000, and 500 m, respectively. Applying this factor is based on an assumption that pixels close to a land cover type are more likely to be converted to that cover type. We also defined the boundary of the protected area as a constraint layer that prevents land conversion inside the protected area. After this, we used the required input data and ran the model to visualize future land cover maps under each of the three scenarios.

2.2.2.2. FRAGSTATS software. After providing the current and plausible future land cover maps through scenario modeling in our study landscape, we applied the FRAGSTATS 4.1 software which is a spatial pattern analysis program for categorical maps representing the landscape mosaic model of landscape structure (Ma et al., 2013; McGarigal and Marks, 1994; Zhang et al., 2015). We computed four landscape metrics: class area – how much of the landscape is comprised of a particular patch (m²), the percentage of landscape – proportional abundance of each patch type in the landscape, the largest patch index – percentage of total landscape area comprised by the largest patch, number of patches of the corresponding patch type, in both levels of land cover classes and the whole of the landscape. Such metrics are appropriate variables for measuring and quantifying different aspects of the landscape visual patterns in a given time (Herold et al., 2002; Narumalani et al., 2004) and they have been applied by many authors for assessing spatial-temporal changes of land cover in different landscapes (de Barros Ferraz et al., 2005; Deng et al., 2009; Herold et al., 2002; Lausch and Herzog, 2002; Seto and Fragkias, 2005; Uuemaa et al., 2013). Further information on the landscape metrics computed using the FRAGSTATS software are provided in Appendix A.

2.2.2.3. Biodiversity: habitat quality model. The InVEST biodiversity model is habitat-oriented and combines land cover data with existing threats to produce a habitat quality map (Sharp, 2014). If changes of habitat are taken as a proxy for changes in genes, species and ecosystems, then the user can assume that an area with high-quality habitat will better support all levels of biodiversity and that areas, where habitat context has been decreasing over time are declining in terms of resilience, biodiversity durability, extent, and depth. To run the model in our study landscape, using a simple binary approach, we assigned a habitat role to different land cover types. All natural covers, including forests, rivers, and bare lands were considered as habitat and received a numeric value of 1 for each cell in a raster layer and all human-managed cover, including farmlands, settlements, roads, and citrus groves were considered as non-habitat with 0 cell value. The basis of this approach is an island-ocean model that assumes that the managed land matrix surrounding remnant patches of unmanaged land is unusable from the standpoint of species (MacArthur, 1967). The

model also requires data on habitat threat density and its effects on habitat quality. Adjacent habitat cover is also threatened since management of anthropogenic land cover types has led to fragmentation and edge effects on habitat parcels facilitating entry of pollutants, invasive species and other deleterious effects. Therefore, we considered all non-habitat cover as sources of threats, assigning a pixel value of 0–1 of which 1 indicates the cells that contain threats and 0 otherwise. Then we considered four important factors which influence the impacts of threats to habitat in a grid cell: the relative impact of each threat; the distance between habitat and the threat source; the level of legal protection from disturbance in each cell; and the relative sensitivity of each habitat type to each threat to the landscape. Input data for the InVEST biodiversity model are summarized in Appendix B.

By calculating the factors in Appendix B, the model computes the total degradation in each cell and translates it to habitat quality value using a half saturation equation. More details on the mathematical equations and calculation methods used and generally how to quantify habitat quality score for each pixel have been outlined previously elsewhere (Bai et al., 2011; Baral et al., 2014a; Leh et al., 2013; Polasky et al., 2011; Tallis et al., 2013).

2.2.2.4. Climate regulation (carbon storage and sequestration). Using the relationship between land cover and carbon sinks in soil and vegetation is the easiest way to estimate the total carbon in the earth system (Nelson et al., 2009). To generate a carbon storage map associated with different types of land cover the InVEST model calculates the total carbon in a land parcel based on aggregating the amount of carbon stored in four sinks: above-ground biomass, below-ground biomass, soil and dead organic matter. Input data for the InVEST Climate Regulation model are provided in Appendix C. Although the ideal method for estimation of carbon stocks is a direct field measurement this was not possible due to time and budget constraints and so we used a set of existing data sources. We estimated the amount of carbon in each of these sinks as a function of land cover distribution and biomass age in the following way. First, the above-ground biomass density was calculated using the FAO method for estimating biomass density of woody formations based on existing forest inventory data (Brown, 1997). With this method, biomass density is simply calculated through multiplication of the average forest stand volume by the average wood density and a biomass expansion factor. Then, based on the ratio of root to shoot, 20% of the above-ground biomass was considered as below-ground biomass in each land cover class (MacDicken, 1997; Mokany et al., 2006). The amount of forest dead wood biomass was also considered equivalent to one-tenth of the above-ground biomass (Delaney et al., 1998). Then, to convert metric tons of biomass to metric tons of carbon, the ratio of 0.49 was used as the conversion factor (IPCC, 2006) (See Appendix D for more details and calculations). To account for carbon stocks stored in the soil, we used existing published data (Haghdooost et al., 2011; Jafari and Mesri, 2015). The amount of sequestered carbon in the landscape was also calculated by subtracting the amount of carbon stored in the land at the beginning and at the end of the targeted time period.

2.2.2.5. Water yield model. Mapping water supply requires models and indicators to estimate the volume of extracted and available water in spatial units such as river basins for human consumption. We applied the InVEST water yield model to map the average annual water yield and to determine the relative contribution of each parcel in the study landscape. The model determines the amount of water running off each pixel of land as the total precipitation less the fraction of the water that undergoes evapotranspiration (Sharp, 2014). Then it calculates the sum and average water yield in sub-watersheds. The preliminary data used to run the water yield model are summarized in Appendix E. Information on rainfall and evapotranspiration were collected from the Iranian meteorological organization database and its synoptic station in Ramsar city which is located in the study landscape. To estimate the average annual evapotranspiration, we used the ‘modified Hargreaves’ equation (Droogers and Allen, 2002; Samani, 2000). This equation is applied to estimate the reference evapotranspiration using minimum climatological data. The ‘modified Hargreaves’ uses the average of the mean daily maximum and mean daily minimum temperatures, the difference between the mean daily maximum and the mean daily minimums and extraterrestrial radiation, all of which were taken from the Ramsar synoptic meteorological station that are recorded in the Iranian Meteorology Organization daily database (<http://irimo.ir/far/>). To provide factors related to soil, including soil depth as a proxy for root restricting layer depth (Sharp, 2014) and plant available water, because these data were not available, we used data from the Harmonized World Soil Database (HWSD) provided by FAO (Nachtergaele et al., 2008). Plant available water content (PAWC) is the fraction of water that stored in the soil profile that is available for use by plants. Plant available water quantity was taken from HWSD and divided by soil depth to obtain the PAWC fraction throughout the study landscape (Sharp, 2014). This was ground-truthed with local data, including topsoil texture available from a report on Glandrood’s forestry plan (Anonymous, 1998) which was the nearest forestry project in our study area in the western part of Mazandaran province. Given that the model assumptions are based on processes understood at the sub-watershed scale, polygon shape files on watersheds and relevant sub-watersheds were provided using the Arc Hydro tool in Arc GIS software and digital elevation model (DEM). As a result, the landscape was divided into eight watersheds and 18 sub-watersheds. In addition to water yield and supply, the changes in water consumption across the landscape which is caused by land cover change was also estimated. Consumptive water use is that part of the water used that is incorporated into products or crops, consumed by humans or livestock, or otherwise removed from the watershed water balance. For farmlands, water used by agricultural processing that is not returned to the watershed was considered. In human settlements, water use was calculated based on an estimated water use per person and multiplied by the approximate population area per raster cell.

3. Results

3.1. Land cover changes under plausible future scenarios

Using the Scenario Generator’s output maps as input layers for the FRAGSTATS tool, it was possible to compare and analyze changes in the current land cover as a consequence of each three plausible scenarios.

Fig. 2 shows that under the BAU scenario agricultural and residential land uses are intensified not only in low-elevation areas of the study landscape, but also are spread as far as the border of the protected area. As a result, significant parts of natural forest cover

Table 2

An estimation of the area covered (ha) by each type of land cover under each scenario.

Land cover	Class area (ha)			
	Current Landscape	BAU	PBZ	CZ
Low-density forest	6558.21	5550.57	5824.98	5827.95
Semi-dense forest	22,934.61	21,600.63	22,549.41	22,006.44
Dense forest	18,622.62	18,488.52	18,605.70	18,527.13
Rivers	443.79	443.79	443.79	443.79
Farmlands	1903.95	2861.91	2336.04	2660.31
Citrus groves	391.86	616.77	631.35	640.80
Water body	12.96	12.96	12.96	12.96
Settlement	1360.53	2548.98	1591.74	2040.75
Roads	1368.36	1436.76	1600.92	1436.76
Bare land	2232.27	2232.27	2232.27	2232.27

will convert to human settlements on upstream land in the external eastern part of the protected area.

In the event of a significant expansion of the protected area, as well as the continual expansion of anthropogenic land uses outside of the protected boundary (PBZ scenario), the spread of human settlements to higher altitudes will be prevented. However, almost all of the remnant patches of natural vegetation downstream will be converted to farmlands and settlements. Finally, under the CZ scenario which is redefining the boundary of the protected area through interaction with local participating stakeholders in environmental protection through controlled land conversion, we observe an increased development of farmlands, settlements, and gardens by 50, 50 and 66%, respectively. Table 2 shows estimates of the class area metric (ha) for each type of land cover under each scenario. Currently, about 9% of the land area is under human use and the remaining 91% is covered by natural forests and river ecosystems. The model outputs show that under the BAU, PBZ and CZ scenarios, 13, 11 and 12%, respectively, of the landscape will be under human use. However, the spatial distribution of these areas will be different for each scenario (Fig. 2).

According to class area and percentage of landscape metrics the dominant cover of the study landscape is semi-dense forest. In terms of visual changes to the landscape the number of patches and the largest patches index (%) are shown in Fig. 3(a and b). While the number of patches of all three classes of the natural forest covers will considerably decrease under BAU and PBZ scenarios, such reductions are much less under CZ scenario (Fig. 3a). For example, the number of the semi-dense forest patches decrease to 645, 590 and 255 respectively under BAU, PBZ, and CZ scenarios. In contrast, farm patches will increase to 227, 164 and 260 respectively under these scenarios. Although the greatest increase in agricultural patches is with the CZ scenario representing a win-win solution for both objectives of development and environmental protection, the least reduction in forest patches is under this scenario which may be interpreted as a better spatial distribution of farmlands. Human settlement patches increase to 94, 173 and 6, respectively, for BAU, PBZ, and CZ scenarios. This is really an interesting result since although the settlement patch numbers are 6 under CZ scenario, the area covered by this land cover class increased by 66% which means this scenario can prevent the sprawling housing across the landscape in comparison with other scenarios. However, the visual changes of the landscape compared to the current state will be much less than in the BAU and PBZ scenarios as can be directly seen in Fig. 2, because some parts will no longer be conservation areas and are less spatially limited for human land use. Therefore, while a significant part of the key social and economic objectives of local stakeholders can be achieved, the intensification of anthropogenic cover downstream or its spread to upstream is prevented.

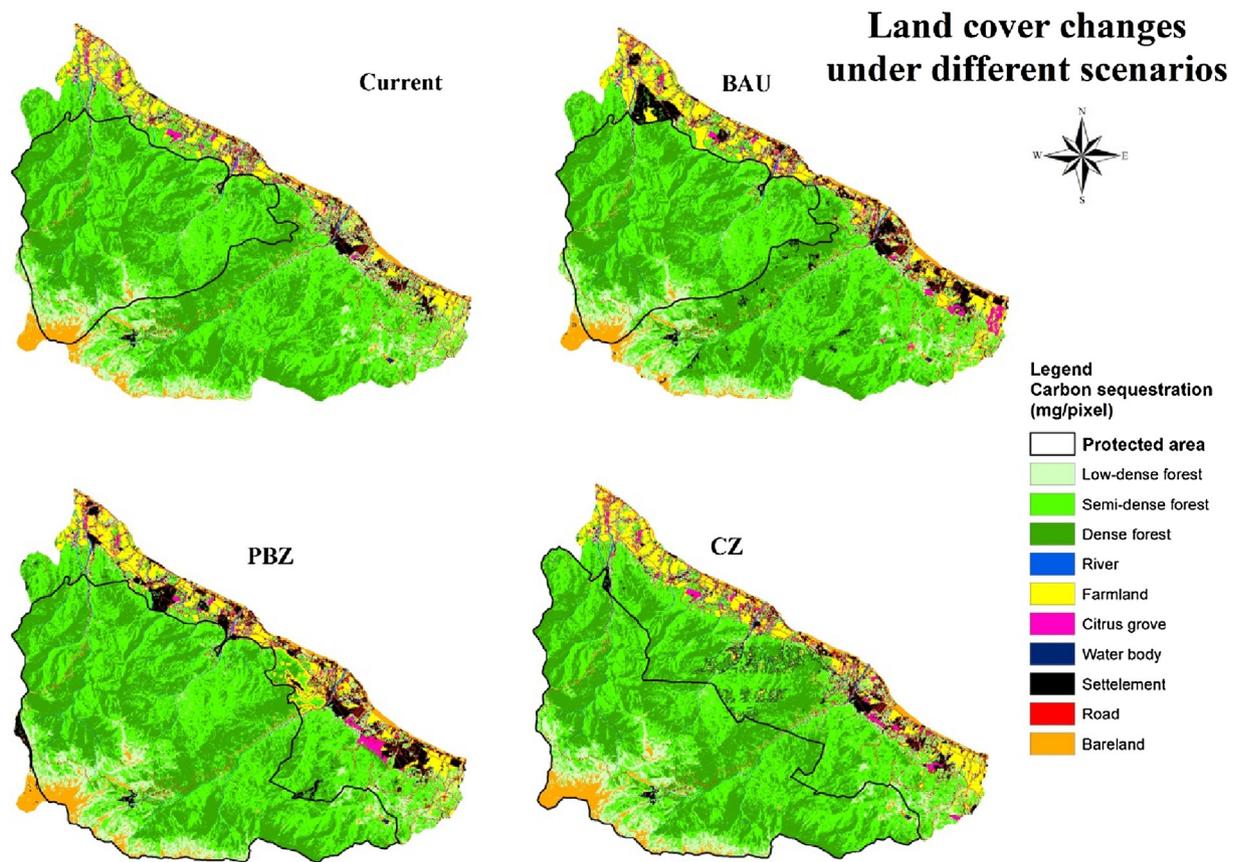


Fig. 2. The InVEST model outputs on land cover changes under different scenarios.

The largest patches of the semi-dense forest class are predicted to dramatically decrease from 28% under current situation to approximately 13% under BAU (which means the less forest cover) and 24% and 21% respectively under PBZ and CZ scenarios (Fig. 3b).

3.2. Changes to ES flows

3.2.1. Biodiversity and habitat quality

The predicted biodiversity maps under each scenario which show relative habitat quality score in a range from zero to one across the whole landscape (Biodiversity Model outputs) are shown in Fig. 4a. The minimum, maximum and average quality scores in the current landscape were 0, 0.99 and 0.30, respectively. High-quality areas are located mainly in the higher elevations and in the southern part of the study area which is at some distances from areas of human land use which might act as threats to biodiversity. Under the BAU, PBZ and CZ scenarios, the average quality of habitat declined by 0.26, 0.23 and 0.18, respectively. The rate of decline in habitat quality and, consequently, land degradation under the CZ scenario is much less than for the BAU and PBZ scenarios. Under the BAU scenario, areas with a high degree of habitat quality and biodiversity are drastically reduced. Under the PBZ scenario, because of the proximity of threat resources to the border of the protected area and the spatial diffusion of these threats, major sections inside the protected zone will be low-quality habitat. Under the CZ scenario, some of the areas with high biodiversity value that are not protected under the current circumstances will fall within the newly-defined protected areas. In addition, we observe the maximum habitat quality distribution inside the new defined protected boundary due to the increasing distance of the threat sources from the borders of the protected area, resulting from better protection

Table 3

The classification of the current and different scenario landscapes relevant to habitat quality scores.

Habitat quality classes	Percentage of Landscape			
	Current	BAU	PBZ	CZ
No quality	9.02	13.45	13.38	12.16
Weak quality	10.02	35.23	13.95	11.73
Moderate quality	38.17	29.26	38.23	38.40
Good quality	25.71	16.96	21.76	22.21
Excellent quality	17.06	5.1	12.65	15.47

zoning. To show a more tangible representation of such changes we categorized the average relative value of habitat quality of landscape under each scenario into five classes of quality (None (0), Weak (0.25), Moderate (0.5), Good (0.75) and Excellent (1)) and calculated the percentage of landscape covered by each class as shown in Table 3.

3.2.2. Carbon storage and sequestration

Based on modeling, 13,444,293 metric tons (Mg) estimated as carbon storage in the current forested landscape of the Sarvelat and Javaherdasht area. The amount of carbon stored will be reduced by 2.32% (13,131,370 Mg); 2.29% (13,135,574 Mg) and 1.44% (13,249,510 Mg), respectively under the BAU, PBZ and CZ scenarios. Fig. 4b compares the changes in the amounts of carbon sequestered in each of the possible future scenarios.

In this map areas where carbon storage does not change due to land cover changes, are indicated by the yellow background and zero change value. Negative values for carbon sequestration are shown in a range of different colors and a range of quantities between 1 and 22 tons per pixel. This reduction in sequestration

Table 4
The effects of changes in precipitation and evapotranspiration on landscape water yield under different future scenarios.

Scenarios	Average Precipitation (mm)	Average Evapotranspiration (mm)	Water yield (mm)		
			Minimum	Maximum	Average
Current	650.71	61.94	0	1310.89	303.59
BAU	650.71	59.61	0	1310.89	305.92
PBZ	650.71	59.72	0	1358.18	305.81
CZ	650.71	60.77	0	1348.57	304.77

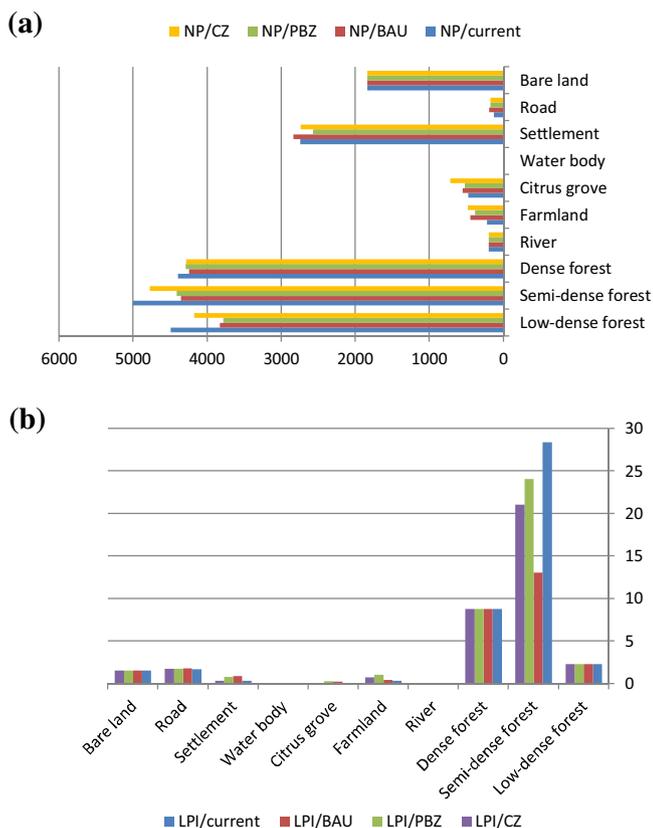


Fig. 3. Number of patches (3a) and the largest patch index (3b) of each class of land cover under current and plausible future scenarios.

potential is mainly due to conversion of natural forest cover to anthropogenic land uses. Small areas of white color indicate positive values maximally up to 2.6 Mg and are largely due to an increase in the area of rice fields that contain higher soil organic carbon.

Given that all three scenarios show a decline in natural forest cover, the changes in carbon sequestration across the landscape are, not surprisingly negative in all three scenarios (−312,922; −308,719 and −194,783 Mg, respectively under the BAU, PBZ and CZ scenarios). This means that the carbon sequestration quantity of the landscape study will be 37% greater under CZ scenario in comparison with BAU scenario, while this amount for PBZ scenario is just 1.34%. The CZ scenario, therefore, appears to be the most climate change efficient policy because of its greater capacity for carbon sequestration over time.

3.2.3. Water-related ES

Water balance raster maps at the pixel scale and a water supply vector maps at the scale of watersheds and sub-watersheds were produced for each of the future scenarios.

Table 5

Quantitative changes of water yield, supply and demand in gegalitres/year under each scenario.

Water-related ES	Scenarios			
	Current	BAU	PBZ	CZ
Water yield	294.19	296.45	296.35	295.33
Water supply	272.18	260.08	260.20	263.78
Water demand	22.00	36.37	36.14	31.55

3.2.3.1. Water balance. Table 4 compares the minimum, maximum and the average water balance in the current landscape with those for each of the possible future scenarios. It shows that the average water balance and the amount of available water will increase under all scenarios because conversion of natural forest cover will result in a reduction in the amount of evapotranspiration. Under the BAU scenario, the maximum amount of tree cover considerably reduced and we would see the greatest increase in the amount of surface runoff which is equal to 2,237,850 m³/year. These changes are minor and would be the least and equal to 1137139 m³/year under the CZ scenario. However, since changes in evapotranspiration are low, and the amount of actual evapotranspiration is always less than the amount of precipitation, it is predicted that these changes will not have severe effects on the water balance. What we have not accounted for is greater water use efficiency by trees with increased CO₂ levels and hence greater runoff since these will increase proportionally under any of the scenarios.

3.2.3.2. Water supply and demand. Table 5 shows the total volume of water yield and the actual supply and demand of water (consumption) under each of the scenarios. Water yield is predicted to increase under all scenarios, but given that the water supply is higher than demand due to rainy conditions prevailing in the region, this increase in water yield will result in an increase in surface runoff. This may result in increased risk of flooding and ultimately means a reduction in ecosystem potential in flood control. Under the CZ scenario, this problem will be reduced since, as we described previously, the changes in surface runoff are not great. Also, the actual supply of water where and when it is needed for human-made land uses will decrease. This is because of the reduction in water supply potential as a result of tree clearing. However, the CZ scenario presents the least reduction in water supply potential which means it is a more water efficient scenario in comparison with the other two scenarios, particularly the BAU scenario. Also, the demand for water will rise proportionally, based on the quantity of forest land converted under the BAU scenario, we would see the greatest increase in water consumption. In the current landscape, watershed numbers 1, 3, and 5 with more than 700 m³/ha and watershed numbers 3 and 7 with more than 100 m³/ha represent those with the highest water supply and demand, respectively. Under the BAU scenario, in addition to the above-mentioned watersheds, water supply will increase to more than 700 m³/ha in watershed number 7 and the demand trend will also rise in watershed numbers 3 and 7, by 71% and 42%, respectively (Fig. 5).

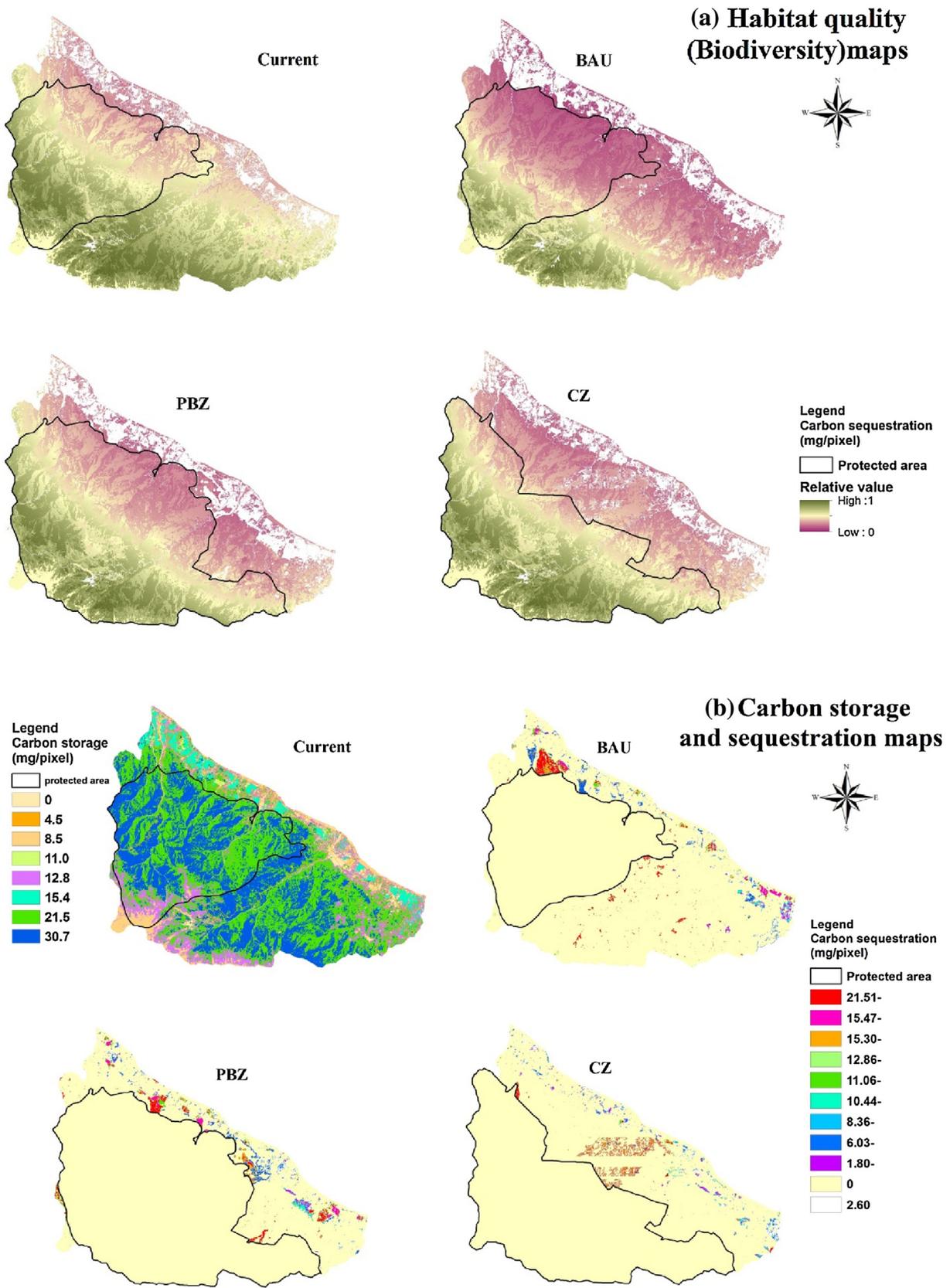


Fig. 4. The InVEST model outputs on habitat quality (biodiversity) (4a) and climate regulation (4b) changes and quantities under current, BAU, PBZ and CZ future plausible scenarios.

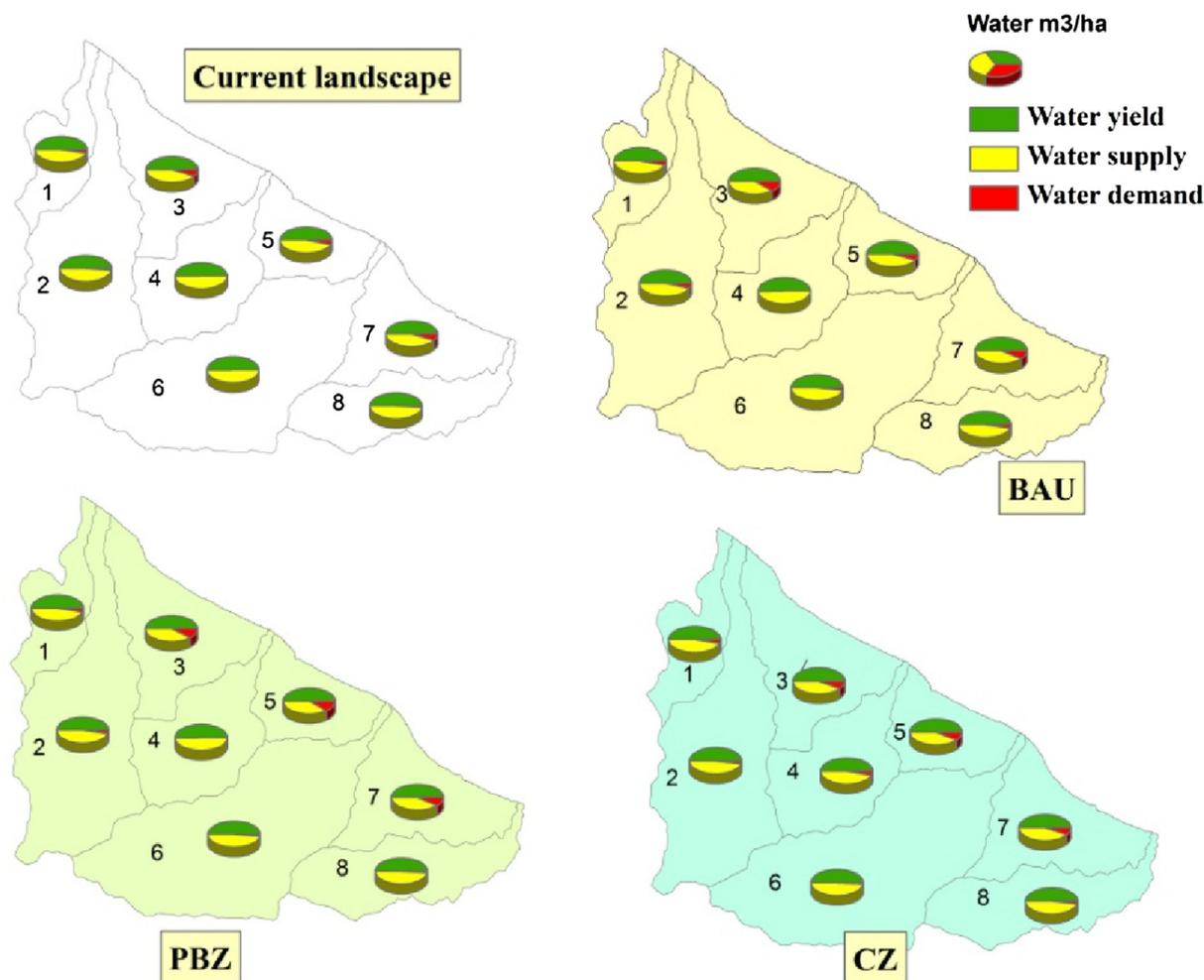


Fig. 5. Changes in water supply and demand (m^3/ha) in each watershed under different scenario.

Under the PBZ scenario, the water supply potential is similar to the current situation, but the water demand in watershed number 5 will increase from 67 to $180 \text{ m}^3/\text{ha}$ due to intensification of farmlands which is a dramatic change. And finally, under the CZ scenario the changes in water supply and demand will be the lowest compared to the current landscape. Under this scenario, the water consumption in watershed 5 will increase to $139 \text{ m}^3/\text{ha}$. In the remaining watersheds, the situation will be similar to the current condition which means the least impacts resulting from land cover changes on water-related ES.

4. Discussion and conclusion

In this paper, interactions between land cover changes and multiple ES supply were modeled under three different land use scenarios. It demonstrated that continuation of the existing trend of land cover change in the absence of a structured land use planning strategy and based on local residents' interests, under the BAU scenario the declining supply of ES will continue to be exacerbated. On the other hand, expansion of the protected area boundaries by the government as a preventive regulatory policy, contrary to local stakeholders' expectations (PBZ scenario), would not ultimately result in an optimal improvement of habitat quality and multiple ES provision. Under this scenario, although the area under protection would expand considerably, the areas covered by good and excellent habitat quality will decline by 4% and 5%, respectively (Table 3) due to the continuing external threats from the

surrounding anthropogenic land uses. It suggests that widening the protected area while ignoring its relationship with surrounding unprotected areas is not an effective environmental policy. This observation is similar to that of a previous study (Palomo et al., 2013). The CZ scenario showed how land use management through the redefinition of the conservation zone and a more efficient allocation of land to human uses, could reduce ecological risks as well as achieve the objectives of both stakeholders concerned with conservation and those concerned with society and development. In addition, the current zoning of the protected area was made based on subjective assessments and expert opinion, not on the basis of measurements of ecological flows. Our modeling shows how we can identify areas that should be protected for their ES that are not currently under protection. In summary, the CZ scenario as a win-win strategy shows that if the zoning of protected area is done based on both potential and realized ES supply and its spatial distribution and is also integrated with other management strategies, including land cover planning outside the protected zone, both objectives of human development and ecological risk reduction could be achieved simultaneously, and thus better decisions could be made.

The scenario building of land cover changes in this study is comparable to those for the previous investigation such as Bhagabati et al. (2012), Goldstein et al. (2012), Baral et al. (2014a). Using ES information Bhagabati et al. (2012) identified the most appropriate policy for sustainable land use planning at the province and district level in Sumatra, Indonesia. They modeled and quantified multiple

ES in a region where forest cover was increasing based on governmental visions of land use future plans. Similarly, in consultation with stakeholders,

Baral et al. (2014a) identified and defined five plausible future land use scenarios in south-eastern Australia. They quantified and valued five important ecosystem goods and services under five future land use scenarios and showed that an ES framework can be used to assess and value different land use options and demonstrated the potential to manage landscapes to produce mix set of ecosystem services. Goldstein and colleagues developed seven scenarios using InVEST models in Hawaii, impacts to ES were assessed and the results were used to make policy choices (Goldstein et al., 2012). Our study highlights a real-world situation of using InVEST scenarios to better inform planning and land use policies.

We used qualitative inputs of local residents interests as inputs for scenario modeling, which is usually a challenge in scenario building. In fact, what distinguishes our scenario modeling from previous vision-based scenario modeling is the possibility of outlining the future land cover based on the involvement of a variety of factors including land suitability, stakeholder's priorities, and drivers of land cover changes. Although this is difficult to do because of the complex interactions between such heterogenous variables, using the InVEST scenario generator simplified this process. However, there are some limitations of applying geo-spatial and remote sensing approaches, including InVEST tools for biodiversity assessments such as, lack of assessment of small-scale characteristics and finer details in many cases (Spanhove et al., 2012) and the need for field verifications for better accuracy (Baral et al., 2014b; Lück-Vogel et al., 2013).

In terms of ES tradeoffs and synergy analysis, we quantified the spatial overlap between habitat quality and carbon storage using correlation coefficients and then positively correlated ES assumed to be synergistic. It was observed that there is a synergy between habitat provision and carbon storage with a correlation coefficient of 0.5034 which is interpreted statistically as a moderated significance linear relationship between them as both provided for through protection of forested habitats and patches. Although the provision of food and provision of ecosystem goods were not assessed individually in this study, agricultural expansion under all scenarios results from forest conversion. Therefore, it could be interpreted that there is a negative correlation and thus a tradeoff between this service and other supporting and regulating services in the landscape as shown previously by others (Baral et al., 2014a; Carreño et al., 2012; Nelson et al., 2009; Polasky et al., 2011; Power, 2010; Raudsepp-Hearne et al., 2010). In addition, the non-deliberate increase of water yield in the form of surface runoff as a consequence of forest conversion provides an example of unusual trade-offs between water supply and other ES in our study area.

In summary, we considered land cover changes as driving forces of change in supply of and threat to ES. However, it should not be forgotten that there are some other very important drivers that may be at play here, such as illegal logging, illegal hunting and pollution. In addition, some indirect drivers, including financial policies in the agricultural sector, market prices, especially with land and housing, cannot be incorporated in the InVEST models. We recognize these as shortcoming of our study. InVEST models, although grounded in theory, offer low-precision estimates of ecosystem service provision and provide coarse assessments of ecosystem services (Tallis et al., 2013). For this reason, the recommendations in this paper should be interpreted as feasibility assessments rather than as detailed guidelines for targeting precise locations of ecosystem service provision as well as site selection for different kinds of man-made development. We encourage further review and validation for new uses of our assumptions and recommendations which could be enhanced through field studies and more detailed fine-scale assessments. Moreover, in this study, socio-economic factors

were not considered in depth and our recommendations serve as a first-cut feasibility analysis based on biophysical assessment of habitat quality and ES. Further consideration of social, economic and governance characteristics ought to be investigated through social-environmental studies. We hope our key findings are useful for spatial planners at the provincial and district levels, as well as government agencies and other institutions that are considering investing in the region. We encourage further review and validation for new uses of our results and recommendations.

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Appendix A.

Table A1

Table A1
Brief information on selected landscape metrics which computed using the FRAGSTATS software.

Metrics	Description	Unit
Total (Class) Area (CA)	Class area is a measure of landscape composition; specifically, how much of the landscape is comprised of a particular patch type.	m ² , ha
Percentage of Landscape (PLAND)	Percentage of landscape quantifies the proportional abundance of each patch type in the landscape.	%
Number of Patches (NP)	NP equals the number of patches of the corresponding patch type (class). It is a simple measure of the extent of subdivision or fragmentation of the patch type.	None
Largest Patch Index (LPI)	Largest patch index at the class level quantifies the percentage of the total landscape area comprised by the largest patch. As such, it is a simple measure of dominance.	%

Appendix B.

Table A2

Table A2
Input data for InVEST biodiversity model.

Input data	Description
Current land cover map	A GIS raster dataset, with a numeric land cover code for each cell
Future land cover map	A GIS raster dataset that represents a future projection of land cover in the landscape (the output of the scenario generator model)
Threat data	A table of all threats the model should consider. The relative impact of each threat is considered using a relative range value of 0–1 where 1 indicates the most intensive threats. In the study area, we set the values of 0.6, 0.7, 1 and 1, respectively, for threats to citrus groves, farmlands, settlements and road networks adjacent to natural habitats.

Table A2 (Continued)

Input data	Description
Sources of threats	Distances between habitats and threat sources were considered as mitigating factors, because, we suggest, the impact of a threat on a habitat decreases across space as distance from the degradation source increases. Thus, it is expected that grid cells that are closer to threats will degrade more because of more intensive impacts. We set this factor equal to 6 and 10 km for citrus groves and farmlands, respectively, and 20 km of settlements and roads which create more degradation based on previous experience in the study area.
Accessibility to sources of degradation	GIS raster file of the distribution and intensity of each individual threat.
Habitat types and sensitivity of habitat types to each threat	A GIS polygon shape file containing data on the relative protection that legal/institutional/social/physical barriers provide against threats.
Half-saturation constant	A table of land cover types, whether or not they are considered habitat, and for land cover types that are habitat, their specific sensitivity to each threat. This factor is considered in the model with a value range of 0–1 where values closer to 1 indicate greater sensitivity.
	The InVEST model uses a half-saturation curve to convert habitat degradation scores to habitat quality scores (Tallis et al., 2013). An inverse relationship between the degradation score and its habitat quality score is determined by this half-saturation constant.

Appendix C.

Table A3

Table A3
Input data for InVEST Climate Regulation model.

Input data	Description
Current land cover map	A GIS raster dataset, with a land cover code for each cell
Carbon pools	A table of land cover classes, containing data on carbon stored in each of the four fundamental pools for each land cover class.
	The amount of carbon stored in aboveground biomass (in Mg ha ⁻¹)
	The amount of carbon stored in belowground biomass (in Mg ha ⁻¹)
	The amount of carbon stored in soil (in Mg ha ⁻¹)
	The amount of carbon stored in dead organic matter (in Mg ha ⁻¹)
Future Scenarios	Three land cover maps for BAU, PBZ and CZ scenarios which enable the model to measure the sequestration rates in the four major carbon pools on the future plausible landscape

Appendix D.

Descriptions of the FAO method for estimating biomass density of woody formations based on existing forest inventory data and its application in the study area.

To calculate aboveground biomass we used the FAO method for estimating biomass density of woody formations based on existing forest inventory data:

$$\text{Aboveground biomass density (t/ha)} = \text{VOB} \times \text{WD} \times \text{BEF} \quad (\text{D.1})$$

Where:

WD = Wood density here is defined as the oven-dry mass per unit of green volume (either tons/m³ or grams/cm³).

BEF = biomass expansion factor (ratio of aboveground oven-dry biomass of trees to oven-dry biomass of inventoried volume)

According to the Glandroud forestry plan which executed by the Iranian Forests, Rangelands, and Watersheds Organization (IFRWO), the nearest plan for our study area, the average forest stands volumes are equal with 154, 385 and 615 m³/ha respectively in the low-density, semi-dense and high-density forest cover areas. The average wood density of the Hyrcanian mixed broadleaf forests in the northern Iran is 0.65 t/m³. The BEF was considered equal to 1.74 for areas that the biomass of inventoried volume is >= 190 t/ha and 2.66 where the mentioned amount is <190 t/ha. In result, the aboveground biomass was calculated as below:

$$\begin{aligned} \text{Aboveground biomass in the low-density forest} \\ = 154 (\text{m}^3/\text{ha}) \times 0.65 (\text{tons}/\text{m}^3) \times 2.66 = 266 \text{ t/ha} \end{aligned}$$

$$\begin{aligned} \text{Aboveground biomass in the Semi-dense forest} \\ = 385 (\text{m}^3/\text{ha}) \times 0.65 (\text{tons}/\text{m}^3) \times 1.74 = 435 \text{ t/ha} \end{aligned}$$

$$\begin{aligned} \text{Aboveground biomass in the dense forest} \\ = 615 (\text{m}^3/\text{ha}) \times 0.65 (\text{tons}/\text{m}^3) \times 1.74 = 695 \text{ t/ha} \end{aligned}$$

$$\begin{aligned} \text{Below-ground biomass in the low-density forest} \\ = 20\% \text{ of aboveground biomass} = 266 \times 20\% = 53 \text{ t/ha} \end{aligned}$$

$$\begin{aligned} \text{Below-ground biomass in the semi-dense forest} &= 435 \times 20\% \\ &= 87 \text{ t/ha} \end{aligned}$$

$$\text{Below-ground biomass in the dense forest} = 695 \times 20\% = 139 \text{ t/ha}$$

$$\begin{aligned} \text{Deadwood biomass in the low-density forest} \\ = 10\% \text{ of aboveground biomass} = 266 \times 10\% = 27 \text{ t/ha} \end{aligned}$$

$$\begin{aligned} \text{Deadwood biomass in the semi-dense forest} &= 435 \times 10\% \\ &= 43.5 \text{ t/ha} \end{aligned}$$

$$\begin{aligned} \text{Deadwood biomass in the semi-dense forest} &= 695 \times 10\% \\ &= 69.5 \text{ t/ha} \end{aligned}$$

To convert metric tons of biomass to metric tons of C, multiply by a conversion factor, which varies typically from 0.43 to 0.51. Conversion factors for different major tree types and climatic regions are listed in Table 4.3 on page 4.48 of the IPCC (2006). We used the ratio of 0.49 which is suitable for the humid sub-tropical forests. Table A4 shows metric tons of C in different forest LU/LC in our study area.

Appendix E.

Table A5

Table A4

Estimated metric tons of carbon in different density forest covers in the study area.

Forest land cover	Metric tons of biomass	Metric tons of carbon
Low-density	Aboveground	266
	Belowground	53
	Deadwood	27
Semi-dense	Aboveground	435
	Belowground	87
	Deadwood	43.5
Dense	Aboveground	695
	Belowground	139
	Deadwood	69.5

Table A5

Input data for InVEST Water yield model.

Input data	Description
land cover map	A GIS raster dataset, with a land cover code for each cell
Precipitation (mm)	A GIS raster dataset with a non-zero value for average annual precipitation for each cell.
Average annual reference evapotranspiration (mm)	A GIS raster dataset, with an annual average evapotranspiration value for each cell.
Root restricting layer depth (mm)	A GIS raster dataset with an average root restricting layer depth value for each cell. Root restricting layer depth is the soil depth at which root penetration is strongly inhibited because of physical or chemical characteristics.
Plant available water (PAWC)	A GIS raster dataset with a plant available water content value for each cell. PAWC is a fraction from 0 to 1.
Watersheds	A shapefile, with one polygon per watershed.
Subwatersheds	A shapefile, with one polygon per subwatershed within the main watersheds specified in the Watersheds shapefile.
Biophysical Table	Tables of land cover classes, containing data on biophysical coefficients used in this tool.
Demand Table	A table of land cover classes, showing consumptive water use for each landuse/landcover type.

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