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FOREST CHANGES DUE TO HUMAN ACTIVITIES IN THE NATIONAL PARK "FRUŠKA GORA" (SERBIA) – ECOLOGICAL AND ECONOMIC INDICATORS

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Abstract – Forest ecosystems are the source of multiple goods and services to humans, the provision of which is deemed to be conditioned by biodiversity. If so, the value of biodiversity has to become apparent to society and, especially, decision makers. The aim of this study was to demonstrate this value. Biodiversity changes were investigated along different stages of forest disturbance in the National Park "Fruška Gora". Both quantitative (structural diversity, species diversity) and qualitative aspects (functional traits of species) have been observed. Parallel analysis of the provision level of certain forest products, resulting from the given forest state, was performed and monetarily expressed. The results indicate that biodiversity lowers with higher disturbance. Moreover, benefits accrue with higher biodiversity. The approach also proved useful in estimating the ecosystem's ability to maintain its functionality; however, further exploration of these links will be needed. This is necessary in future work if biodiversity conservation incentives are to be created.

Key words: biodiversity, forest ecosystems, ecosystem services, economic valuation

INTRODUCTION

Forest ecosystems provide a variety of goods and services to humans, as for example timber, medicinal herbs, recreational opportunities, carbon storage etc. Forest policy in the world, however, has for long time been determined by commercial timber interests, that have contributed to the decline of many other forest services. The conventional logging regimes are serious generators of biodiversity diminution at the genetic, species and community levels (Putz et al., 2000). It is now increasingly recognized that such policy is not ecologically, or economically sustainable, and that biological diversity is of critical importance for the sustenance of ecosystem services (Hooper et al., 2005).

The link between ecological sustainability and human benefits has been recently captured by the concept of ecosystem services (Daily, 1997; MA, 2005). Ecological sustainability relates to the maintenance of ecosystem functions – a subset of ecosystem structures and ecological processes – that underpin the provision of ecosystem services (de Groot et al., 2002), the foundation for economic activity and human well-being (Perrings et al., 1995). Subsequently, the relationship between biodiversity and ecosystem functioning has gained profound scientific interest (see Hooper et al., 2005).

Biodiversity encompasses a variety of scales, from diversity of genotypes to biome dispersal (Wilson, 1992). Still, biodiversity is more often associated with species diversity (Magurran, 1988). It is the importance of this biodiversity component for ecosystem functioning that has been widely used as an argument for biodiversity conservation, while the empirical evidence was rather equivocal (Schwartz et al., 2000). Studies now suggest that the effects on ecosystem processes should be rather attributed to functional diversity, the presence and abundance of certain functional traits of individual species and their interactions. Moreover, functional diversity, as well as functional redundancy is of high importance for the ecosystem's capacity to maintain its functionality under various environmental conditions (Chapin et al., 2000; Díaz and Cabido, 2001).

If forest biodiversity is considered, another component is pulled in, that of structural complexity, which together with composition and function affects ecosystem processes. This refers to the physical organization of the ecosystem, from habitat to landscape level (Franklin, 1988). Therefore, the broad spectrum of structural attributes can be observed, and the choice largely depends on the scale of interest (Franklin et al., 2002). For example, if observed at the forest stand scale, structure concerns the interaction of different stand structural attributes such as variation in tree size, density of large trees etc. (e.g. Spies and Franklin, 1991).

It was suggested that focusing on the stand scale has a higher chance to speed up the process of refining forest management practice, since management decisions are usually applied at this scale (O'Hara, 1998). Still, it is questionable if this type of information alone is a strong enough "weapon" to influence decisions at the operational level. As mentioned at the beginning of the article, financial incentives determine management practice to a great degree. In the world of economics, which relied *a priori* on the commercial use of timber resources, incentives detrimental to biodiversity consolidated. It is now necessary to create incentives that will ensure biodiversity conservation and thus sustain the flow of ecosystem services (Folke et al., 1996; Pearce, 1998).

The notion that biodiversity changes can affect the capacity of an ecosystem to provide ecosystem goods and services has to be empirically supported. On the other hand, this has to be translated into a language understandable to society and especially to policy makers. In the latter case, economic valuation can play an important role by making transparent the fact that biodiversity depreciation bears a cost to humanity (Brander et al. 2010).

Economic valuation of biodiversity, however, is still faced with many complications (Turner et al., 2003; Brander et al. 2010). Biodiversity *per se* would be extremely difficult to value, and many alternative approaches and methods have been developed, where the choice depends on the rationale behind the valuation (SCBD, 2001). If the aim is to demonstrate biodiversity value, an effective approach can be to show the change in its value along different stages of ecological disturbance at a particular location (Turner et al., 2003).

The problem with the estimation of biodiversity value is that biodiversity represents the basis for most of the other ecosystem services (de Groot et al., 2002). In this sense, biodiversity is priceless. However, two important roles of biodiversity can be delineated, which can be a helpful starting point. On the one hand is its role in supporting the provision of ecosystem services, from which the direct or indirect benefits of such services can be calculated. On the other hand is its role in ecosystem resilience – its so-called "insurance value" (Turner et al., 2003). Here, it is perceived that biodiversity loss can result in the loss of ecosystem functionality and, consequently, undermine further provision of ecosystem services.

The aim of this study was to demonstrate the value of biodiversity by depicting changes at the provision level of two types of forest products, resulting from forest structure and composition changes due to different levels of human disturbance. For this purpose, forest stands at different levels of degradation were examined in the National Park "Fruška Gora" in Serbia. The examined forest products were products with actual market value: timber and non-timber forest products (medicinal plants and fruits).

MATERIALS AND METHODS

Fruška Gora is the Pannonian island mountain in

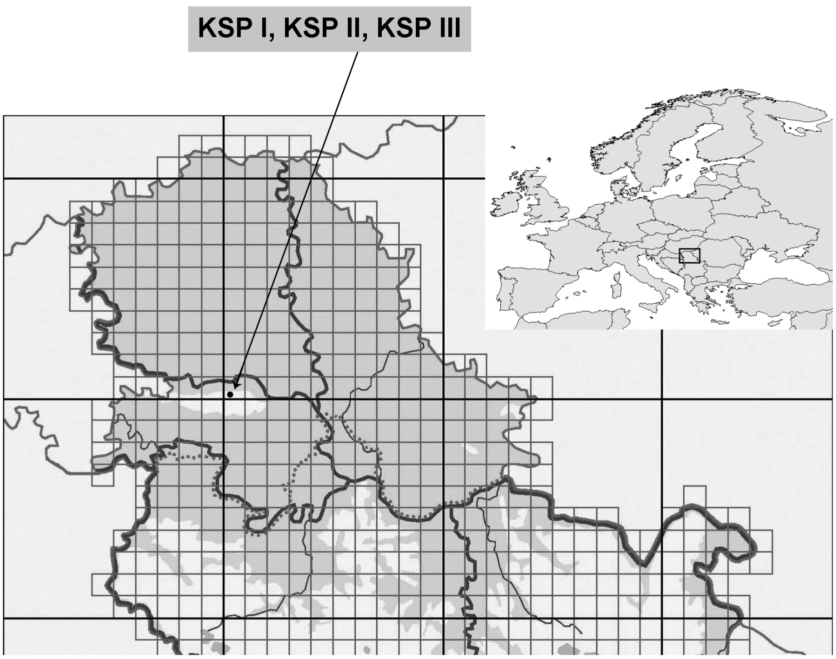

Fig. 1. Map showing the location of three research plots in National Park “Fruška Gora” (Serbia).

northern Serbia (Vojvodina), with the highest peak at 539 m; the south of the Pannonian Plain of central Europe (Fig. 1). Its location, specific geological history and variety of microclimatic conditions (from lowland semiarid climate to the relatively cold and humid submontane climate) make this mountain rich in biodiversity.

The national park, covering an area of more than 25 000 ha, was proclaimed in 1960. From 2008, it has been the site for Long Term Ecosystem Research (LTER). More than 90% of the area is covered by deciduous forests at elevations above 300 m, mainly with mixed stands and the following dominant species: *Quercus petraea*, *Tilia* spp., *Carpinus betulus* and *Fagus moesiaca*. However, more than 80% of the forest is of coppice origin, as a result of severe forest exploitation in the past; clear cuts and regrowth from stumps (Prostorni plan područja posebne namene Fruške gore do 2022. godine, 2004).

The responsibility for forest management was given to public enterprise (Forest Law from 1991). According to the Spatial Plan for the National Park “Fruška Gora” from 2004, a three-level protection regime was established within the area of the national park, delineating: 3.7% to the first (highest) level of protection – deemed for special purposes (non-managed forests), 67% to the second level of protection, and 29.3% to the lowest (third level) of protection; the last two being the managed forests.

The existing division on multiple protection regimes served as the basis for choosing sites for the research. Three research plots of 1 ha were delineated, each being in a different protection regime. Each protection regime is representative of a different stage of forest degradation, where the first regime of protection encompasses the least degraded forests, and the third the most degraded.

For the sake of comparison, it was necessary to include stands with similar floristic composition. Firstly, the inventory for national park forest management (2006) was consulted. The inventory offers basic information on forest stands within a given

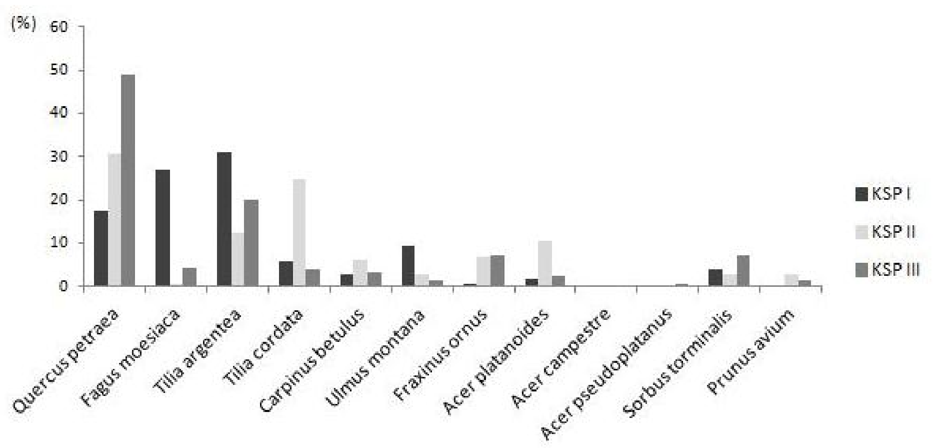

Fig. 2. Percentage of tree species recorded in each research plot.

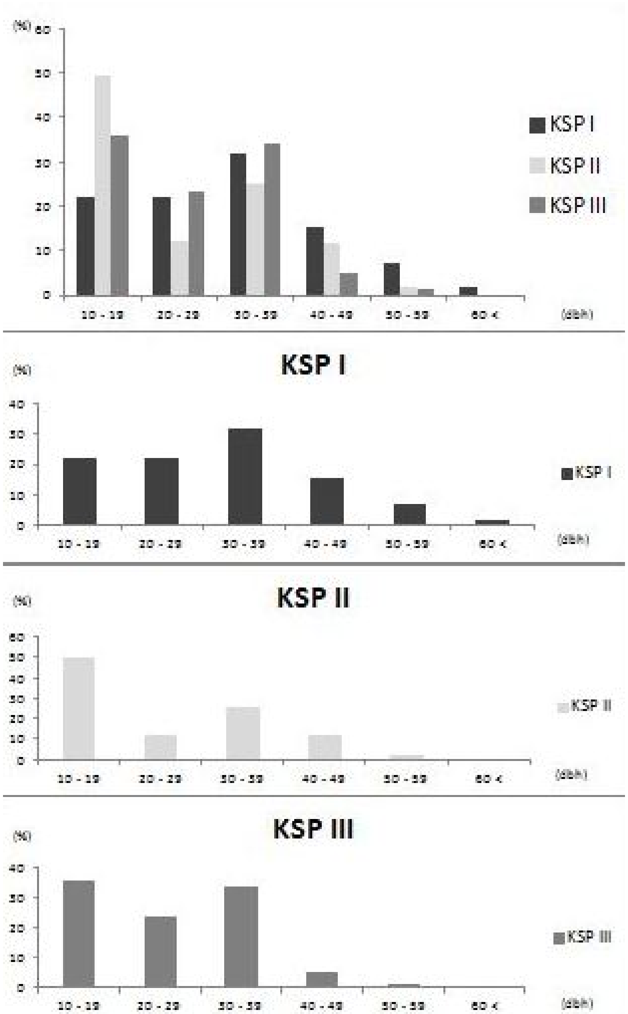

Fig. 3. Percentage of trees in different diameter classes in each research plot.

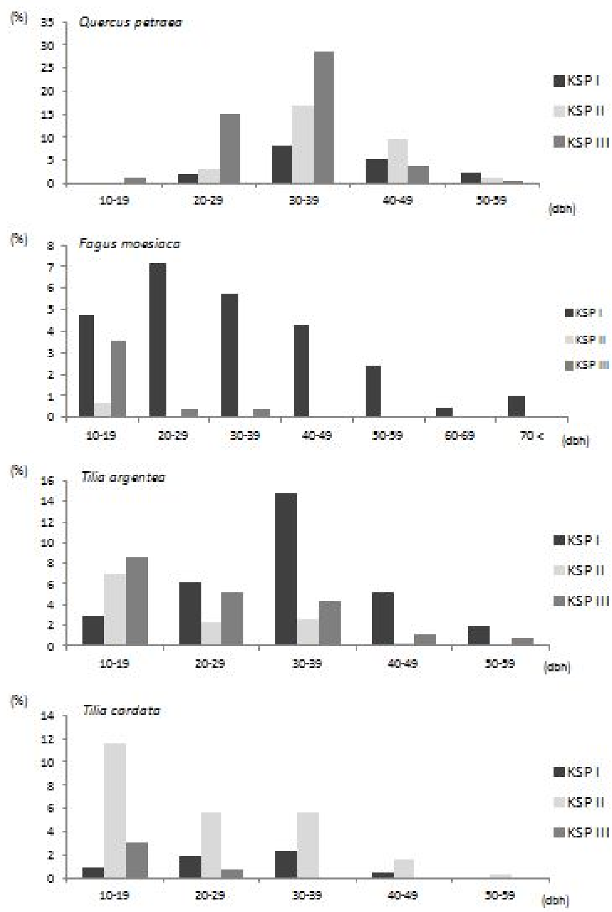

Fig. 4. Percentage of dominant tree species in different diameter classes in each research plot.

management unit (dominant tree species, stand age, tree volume per ha, etc.). The management units are of different size, and the given information relates to a part (1 ha) of the total size of one section within the management unit. The forest inventory, however, does not contain information on species in the herbaceous layer.

In order to increase the probability of choosing similar forest stands in the field, we decided that the management sections be similar in dominant tree species, of similar age and positioned close to each other, but still belonging to different protection regimes. The research plots were designated in a way to keep biophysical factors constant as much as it was possible. All the plots had the same northwest exposure, occupied the same elevation zone (above 400 m), and had a slope between 25-30%. The research plots (Fig. 1) were chosen in the management unit "Popovica-Majdan-Zmajevac" (3804), within walking distance, and were in the shape of a square (100 m x 100 m).

The fieldwork was done at the beginning of June 2012, during maximum growth of the herbaceous layer. The inventory of plants was done along 16 transects (25 m x 25 m) in each research plot. Species were identified in all forest layers: trees, shrubs, and herbs. In the overstory, the following stand structural parameters were measured for each species: tree height and diameter at breast height (dbh at 130 cm) for all trees with dbh ≥ 10 cm. In the understory layer, species were described by using Braun-Blanquet cover-abundance values (Braun-Blanquet, 1964). The scores obtained for species abundance in 16 transects of each plot were summed and averaged. In addition, the presence of dead wood and stumps was recorded, while canopy cover and soil characteristics were visually estimated and described.

To quantify species diversity in the herbaceous layer, the commonly used Shannon index (Shannon and Weaver, 1949), combining evenness and richness, was employed:

$$H' = -\sum_{i=1}^{s} pi \ln pi$$

where *pi* represents the proportion of individuals in the i^{th} species, and *S* is the number of species. The value of the index increases as the number of species increases, and as the species become more equal in abundance.

For the overstory, the index of stand structural diversity (Staudhammer and LeMay, 2001) was applied. The index equals the average score of the following three attributes: tree diameter, tree height and tree species:

H'_{d+h+s} – Shannon's index extended to diameter, height and species (post-hoc method)

where H'_d is Shannon's index for tree diameters, H'_h Shannon's index for tree heights, and H's Shannon's index for tree species. Hence, each component (H'_d, H'_h, H'_s) can be also studied separately.

Prior grouping of tree heights and diameters into classes had to be made. Tree diameters were divided into 10 cm diameter classes, starting from 10 cm. Tree heights were divided into 3 classes (I – trees taller than 20 m, II – trees tall 10 to 20 m, III – trees shorter than 10 m).

The extended Shannon index measures richness and evenness of stand structural diversity. Its advantage is that it is independent from tree size, since it gives equal weight to horizontal, vertical, and species diversity. The higher value of the index is obtained when the stand is more structurally diverse and when distributions are even.

Finally, the obtained data on tree heights and dbh ≥ 10 cm was used to estimate merchantable timber volume. The total timber value per hectare was calculated by using the prices given on the official website of the Serbian Forest Public Enterprise for 2012. The costs of exploitation were estimated by using information from public calls of national parks in Serbia for this service (found on the internet). These costs were deducted from the timber price in order to obtain stumpage value. The production rate for non-timber forest products (NTFP) was estimated

Table 1. Number of trees per species within different diameter and height classes in three research plots.

Species Latin Name	KSP I					KSP II					KSP III				
	DBH class	Height class			Total	DBH class	Height class			Total	DBH class	Height class			Total
		I	II	III			I	II	III			I	II	III	
Quercus petraea	10-19	0	0	0	0	10-19	0	0	0	0	10-19	0	2	1	3
	20-29	0	4	0	4	20-29	0	9	0	9	20-29	2	28	8	38
	30-39	5	12	0	17	30-39	8	43	0	51	30-39	6	55	12	73
	40-49	5	6	0	11	40-49	10	18	1	29	40-49	4	6	0	10
	50-59	3	2	0	5	50-59	3	1	0	4	50-59	0	1	0	1
Total		13	24	0	37		21	71	1	93		12	92	21	125
Fagus moesiaca	10-19	0	0	10	10	10-19	0	0	2	2	10-19	0	1	8	9
	20-29	0	10	5	15	20-29	0	0	0	0	20-29	0	1	0	1
	30-39	3	11	0	12	30-39	0	0	0	0	30-39	0	1	0	1
	40-49	4	5	0	9	40-49	0	0	0	0	40-49	0	0	0	0
	50-59	5	0	0	5	50-59	0	0	0	0	50-59	0	0	0	0
	60-69	1	0	0	1	60-69	0	0	0	0	60-69	0	0	0	0
	70 <	2	1	0	2	70 <	0	0	0	0	70 <	0	0	0	0
Total		15	27	15	57		0	0	2	2		0	3	8	11
Tilia argentea	10-19	0	1	5	6	10-19	0	2	19	21	10-19	0	4	18	22
	20-29	0	6	7	13	20-29	0	6	1	7	20-29	1	7	5	13
	30-39	7	24	0	31	30-39	0	8	0	8	30-39	1	9	1	11
	40-49	3	8	0	11	40-49	0	1	0	1	40-49	1	2	0	3
	50-59	2	2	0	4	50-59	0	0	0	0	50-59	1	1	0	2
Total		12	41	12	65		0	17	20	37		4	23	24	51
Tilia cordata	10-19	0	0	2	2	10-19	0	4	31	35	10-19	0	1	7	8
	20-29	0	3	1	4	20-29	0	10	7	17	20-29	0	0	2	2
	30-39	0	5	0	5	30-39	1	12	4	17	30-39	0	0	0	0
	40-49	0	1	0	1	40-49	3	2	0	5	40-49	0	0	0	0
	50-59	0	0	0	0	50-59	1	0	0	1	50-59	0	0	0	0
Total		0	9	3	12		5	28	42	75		0	1	9	10
Carpinus betulus	10-19	0	0	1	1	10-19	0	0	17	17	10-19	0	0	3	3
	20-29	0	3	2	5	20-29	0	1	1	2	20-29	0	2	1	3
	30-39	0	0	0	0	30-39	0	0	0	0	30-39	0	2	0	2
		0	3	3	6		0	1	18	19		0	4	4	8
Acer platanoides	10-19	0	0	2	2	10-19	0	2	29	31	10-19	0	0	6	6
	20-29	0	2	0	2	20-29	1	0	0	1	20-29	0	0	0	0
Total		0	2	2	4		1	2	29	32		0	0	6	6
Acer campestre	10-19	0	0	0	0	10-19	0	0	1	1	10-19	0	0	0	0
Total		0	0	0	0		0	0	1	1		0	0	0	0
Acer pseudoplatanus	10-19	0	0	0	0	10-19	0	0	0	0	10-19	0	0	2	2
Total		0	0	0	0		0	0	0	0		0	0	2	2
Fraxinus ornus	10-19	0	0	1	1	10-19	0	1	18	19	10-19	0	1	17	18
	20-29	0	0	0	0	20-29	0	1	0	1	20-29	0	0	0	0
Total		0	0	1	1		0	2	18	20		0	1	17	18
Ulmus montana	10-19	0	1	16	17	10-19	0	0	8	8	10-19	0	1	2	3
	20-29	0	2	1	3	20-29	0	0	0	0	10-19	0	0	0	0
Total		0	3	17	20		0	0	8	8		0	1	2	3
Sorbus torminalis	10-19	0	0	7	7	10-19	0	0	8	8	10-19	0	0	15	15
	20-29	0	0	0	0	20-29	0	0	0	0	20-29	0	1	2	3
	50-59	0	1	0	1	50-59	0	0	0	0	50-59	0	0	0	0
Total		0	1	7	8		0	0	8	8		0	1	17	18
Prunus avium	10-19	0	0	0	0	10-19	0	1	7	8	10-19	0	0	3	3
Total		0	0	0	0		0	1	7	8		0	0	3	3
		40	110	60	210		27	122	154	303		16	126	113	255

Table 2. Diversity indexes of two forest layers in three research plots.

Forest layer		KSP I	KSP II	KSP III	Diversity index
Overstory	Tree species	1,67	1,88	1,60	Shannon's index
	Diameter at breast height	1,58	1,31	1,28	Shannon's index
	Tree height	1,00	0,93	0,88	Shannon's index
	Stand structural complexity	1,42	1,37	1,26	Extended Shannon's index
Herbaceous layer		2,76	2,70	2,50	Shannon's index

according to the cover values, since we could not apply any kind of destructive methods in the national park. The prices were obtained from the official website of the Ministry of Natural Resources, Mining and Spatial Planning, for 2012. The costs of NTFP collection were taken as minimal wage rate in Serbia, since these costs are not as high as in the case of timber extraction, and the collection is usually performed by a low-income group of people. All the prices were given in local units (dinar) and were converted to euro using the exchange rate of the National Bank of Serbia for 02.08.2012.

RESULTS

Three research plots were chosen: KSP I – forest in the first level of protection, KSP II – forest in the second level of protection, and KSP III – forest in the third level of protection.

In KSP I, the dominant tree species were beech (*Fagus moesiaca*), sessile oak (*Quercus petraea*), and lime species (*Tilia* spp.). In KSP II and KSP III, beech was almost absent, while sessile oak and lime species were dominant (Fig. 2). The percentage of trees with larger diameter (dbh ≥ 40 cm) was highest in KSP I, while in KSP II and KSP III, the highest percent of trees was recorded in the lower diameter classes (Fig. 3). The highest number of trees within the height class I was in KSP I, as well as the lowest number of trees within height class III. In KSP II and KSP III, the number of trees within height class III was much higher. The number of trees within height class II was comparable between the research plots (Table 1).

Among dominant tree species with dbh ≥ 40 cm, beech occurred in the highest number in KSP I, with some very large trees, while sessile oak in this size was most present in KSP II and KSP III (Fig. 4). In the middle diameter class (30 ≤ dbh ≤ 39), sessile oak made the highest number in KSP II and KSP III, while *Tilia argentea* was the highest in KSP I. In the lower diameter classes, in KSP I beech made the highest percent, in KSP II sessile oak with a comparably high presence of silver lime, while in KSP III, the species of lime were prevalent. Generally, the lime species were most abundant in the lower diameter classes, in all three research plots.

Additional aspects, visually observed, were: the highest presence of dead wood in KSP I, high number of stumps in KSP II and KSP III, higher presence of trees regenerated from stumps and stump sprouts in KSP II and KSP III, and more open canopy cover in KSP II and KSP III. The soil conditions worsened going from KSP I to KSP III; in KSP III with humus layer almost absent and very sparse herbaceous layer.

The value of Shannon's index extended to tree diameter, height and species decreased from KSP I to KSP III (Table 2). Viewed separately, the Shannon index for tree species was highest in KSP II and the lowest in KSP III, while tree diameters and tree heights decreased from KSP I to KSP III. Since it was taken into consideration that changes in the class boundaries may influence the value of the index (Staudhammer and LeMay, 2001), we did another calculation for the 5 cm-diameter classes, but there were no significant changes in the index value. The Shannon index for species diversity in the herb layer decreased from KSP I towards KSP III.

Table 3. Abundance of species in forest understory in three research plots. Shadowed rows present the species with the actual market value. Species marked with asterisk are species valuable for their medicinal properties, but without market value.

Species Latin name	**I**	II	III
Herbs			
*Arum maculatum**	1.09	0.06	0
Asarum europaeum	2.78	0.42	0
Campanula persicifolia	0	0	0
*Campanula trachelium**	0	0.26	0
Cardamine bulbifera	2.84	0.53	0.23
*Chelidonium majus**	0	0	0.03
Daphne laureola	0.11	0.03	0
Dryopteris filix mas	0.02	0	0.03
*Euphorbia amygdaloides**	1.11	0.56	0.13
Festuca montana	0	0.9	0.53
Festuca gigantea	1.3	0	0
Festuca sylvatica	0	0.53	0.56
Galium aparine	0.04	0.8	0.1
*Galium odoratum**	2.01	0.4	0.03
Geranium robertianum	2.9	2.5	0.5
*Geum urbanum**	0	0.06	0
Glechoma hirsuta	0	0	0.2
*Helleborus odorus**	1.89	1.5	0.63
Hepatica triloba	0	0.04	0
Heracleum sphondylium	0	0.03	0
Lamium maculatum	4.64	0.1	0
Lathyrus niger	0	0.1	0.33
Lathyrus vernus	1.26	1.19	0.16
*Lilium martagon**	1.21	0.42	0.03
Mercurialis perennis	2.77	0.8	0.03
Polygonatum latifolium	1.09	0	0
Polygonatum officinale	1.32	0.03	0.06
Pulmonaria officinalis	1.21	0	0.1
Ruscus aculeatus	2.78	0.1	0.2
Ruscus hypoglossum	3.94	1.42	0.6
Stachys sylvatica	0	0.2	0.1
*Urtica dioica**	0.02	0	0
*Viola odorata**	0.09	0	0.23
Climbers			
Hedera helix	5.45	2.84	1.16
*Tamus communis**	0.04	1.7	1.93
Shrubs			
*Cornus sanguinea**	1.26	0.33	0.36
Crataegus monogyna	0.07	0.73	0.4
Rosa canina	0.02	0.5	0.43
Rubus hirtus	4.92	1.73	0.46
Sambucus nigra	1.9	0.92	0.36
Staphylea pinnata	1.38	0.3	1.23

Table 4. The economic value of timber and non-timber forest products in three research plots.

Ecosystem's product	KSP I		KSP II		KSP III	
Timber (species name)	Wood volume (m^3/ha)	Price/ha (stumpage value) (€)	Wood volume (m^3/ha)	Price/ha (stumpage value) (€)	Wood volume (m^3/ha)	Price/ha (stumpage value) (€)
Quercus petraea	51,71	7101.53	75,22	8693,20	72,75	5893,91
Fagus moesiaca	59,03	3593,35	0,00	0,00	0,00	0,00
Tilia spp.	71,46	5053,74	33,61	1874,15	18,00	974,14
Carpinus betulus	2,17	65,82	0,61	16,03	2,73	82,00
Total	184,37	15.814,43	109,44	10.583,37	93,47	6.950,06
Medicinal plants and fruits	Total value (€)		Total value (€)		Total value (€)	
	34,53		6,32		1,76	
Total value for products (€)	15.848,96		10.589,69		6.951,82	

Concerning commercial use, all tree species, except *Ulmus montana*, are commercial species, sessile oak being the most economically valuable. From NTFP with an actual market value, five medicinal plants were found, four plants valuable for their fruits, and one for decorative purposes (Table 3). An additional 12 species are considered as valuable for their medicinal properties, but lacking market value, they were not included in the calculation. The estimated commercial value for both timber and non-timber forest products was highest in KSP I and the lowest in KSP III (Table 4).

DISCUSSION

The tree species composition and the dominance of certain species in the herbaceous layer, as for example *Ruscus aculeatus* and *Ruscus hypoglossum* in all three research plots, indicated the presence of a *Querceto-Carpinetum serbicum* association. The association was found in transition with other forest communities (beech forests with lime or sessile oak forests), aggregating species and elements from the different forest communities, due to specific temperate micro (and meso) climatic conditions created at the connection of different forest communities (Janković and Mišić, 1980). In KSP I, beech was the most abundant species. In KSP II and KSP III the most abundant was sessile oak, but the presence of hornbeam, tree seedlings of beech, and the occurrence of a number of mesophytic species in the understory (e.g. *Ruscus hypoglossum, Arum maculatum*, *Asarum europaeum*, *Euphorbia amygdaloides*, *Cardamine bulbifera*, *Mercurialis perennis*) could indicate a transition habitat.

Janković and Mišić (1980) noted that the presence of remnants of hornbeam and mesophytic elements in places where sessile oak has stabilized with a great number of trees can indicate a degradation stage of the initially present association of sessile oak with hornbeam. In such forests, a more open canopy cover enables higher penetration of species, for example *Rubus hirtus*, *Geranium robertianum*, and *Helleborus odorus*, in KSP II and KSP III the most abundant species, besides *Ruscus hypoglossum*. Consequently, it was concluded that KSP II and KSP III represent a degradation stage of the mentioned association – the second degradation stage – where beech species are co-affected. The same authors delineated five degradation stages of the association, suggesting that reversals between these stages are possible due to the high resilience of the forest association.

The importance of recognizing the existence of multiple states, and the capacity of the ecosystem to reorganize was addressed by Holling (1973) and is a notion that gained the attention of, and has been followed by many other scientists (see Scheffer and Carpenter, 2003). In a system with an arrangement of different states to which the same rules on function apply (e.g. structure and feedbacks), the different states are of the same regime. If these rules differ, then the different states belong to different regimes (Walker et al., 2006). Therefore, if reversals between the degradation stages described by Janković and Mišić (1980) are possible, it means that they represent different states of the same regime, where the difference between the states is observed in the gradual disappearance of otherwise dominant species and a change of forest physiognomy, but the functionality is being kept. However, the system is able to keep its functionality under different disturbances (in this case, mainly human intervention) up to a certain point – threshold – when a shift to another regime commences.

The level of disturbance (gradual or sudden) needed to trigger a shift from one regime to another is the measure of the system's resilience in a given regime, i.e. the capacity of maintaining its functionality (Walker and Meyers, 2004). It is important to recognize that the shifts may bring about large changes, positive or negative, in terms of both ecosystem well-being and human well-being (Walker et al., 2006).

Cumulative pressures (e.g. repeated exploitations) lower the system's resilience and increase the probability of reaching the threshold (Scheffer et al., 2001). The knowledge on thresholds is scarce and they are usually hard to identify, but the distance to a threshold can be signaled, so ecological indicators that can send the signal at an appropriate time are needed (Contamin and Ellison, 2009).

In the case of the examined research plots, gradual changes in biodiversity components across the plots could be observed, as for example, the decrease in species relative abundance (from KSP I to KSP III). The Shannon index for tree species (H's) showed the highest value in KSP II, but this could be explained by the higher number of young stems, characteristic for logged and secondary forests in which reproduction and coppicing of the seedlings takes place due to improved light conditions; such forests often have a higher value of species richness (Uuttera et al. 2000). This is why the inclusion of additional indices (H'_d and H'_h) was an added value, which showed a more even distribution of tree diameters and tree heights in KSP I than in the other two plots (Fig. 3). This is preferable in terms of the overall species diversity (Pielou 1975; in Staudhammer and LeMay, 2001). The qualitative aspect of the presence of rare species in KSP I (*Lilium martagon, Daphne laureola*) supplemented the quantitative observation.

The effects of species composition and their interactions were described by Berger and Puettmann (2004). The change to the almost pure sessile oak forest with a more open canopy cover (in KSP II and KSP III) brought about the change towards xerothermic micro- and meso-climatic conditions, resulting in a lower abundance of mesophytic species, but also enabling the faster growth of silver lime, which is a fast-growing species and a strong competitor for soil nutrients. The occurrence of *Fraxinus ornus* is proof of worsened soil conditions. Hence, the coincidence of functional effect and functional response types was observed, which is important to consider in terms of long-term ecosystem functionality (Díaz and Cabido, 2001).

Certain species have a stronger effect on ecosystem processes than others, for example, by altering abiotic conditions (Hooper and Vitousek, 1997). With regard to this, some species may be more and some less desirable. For example, the dominance of fast-growing plants can bring about high resilience, but low resistance to a community (Lepš et al., 1982). High resilience relates to quick regeneration, but it can make it difficult and costly to return a forest to a previous, more diverse state. The low resistance relates to the lowered ability of the community to resist environmental perturbations, due to the suppression of other species carrying such characteristics. *Tilia argentea* is currently one of the dominant species in the National Park "Fruška Gora", covering 37.6% of the total forest area, but it was much less represented in the past. Its current dominance is explained as being the heritage of severe forest exploitation in the past that enabled it to secondarily invade areas cleaned of forest, disabling renewal of other slower growing species (Prostorni plan područja posebne namene Fruške gore do 2022. godine, 2004). This is an example of how short-term profitability can bring high costs to future generations.

What future value would rise from the current state of the forests in Fruška Gora is under question, as this depends on successional processes, determined by complex species interactions, but which result from the forest management applied. Species evenness deserves higher attention because of its more rapid response to human impact (Chapin et al., 2000). Still, if certain functional types of species influence this component, then it may be more reasonable to invest time in exploring these links, important in terms of practical forest management.

The observed changes in biodiversity components were accompanied by lowered provision of both types of forest products, expressed as their lowered economic value per hectare. This indicates that the production level is dependent on the ecosystem state, and that higher benefits accrue where biodiversity is higher.

The economic value of the NTFP showed to be significantly lower than for timber. However, it should be noted that NTFP can be collected on an annual basis, while trees can be harvested only in certain temporal intervals (approx. 80 years). Calculating for future revenues from the products separately may give a different view on this value, and only the states as found in KSP I can support the future flow, since some minimal safety standard is required for the collection of NTFP. Moreover, a number of other plants known for their healing characteristics were found, but they were not accounted for, due to their market absence. It has often been argued that the problem of changing forestry practice to a more sustainable one lies in the fact that the benefits from NTFP products cannot compete with the profit obtained from conventional logging (SCBD, 2001). However, if the insurance value of biodiversity is considered, the view on more desirable practice can change.

The aggregated value of the benefits rising from the given ecosystem state is only one aspect of the ecosystem's value. Another aspect concerns the ecosystem's capacity to sustain this value under different disturbances – the insurance value (Brander et al., 2010). The states as found in KSP II and KSP III may be closer to the threshold, so their insurance value may be also lower. This is important not to be ignored, since small human impact may trigger large changes in the provision level when the system's distance to the threshold is close (Walker et al., 2009). Here, it should be stressed that the larger portion of the forests in the national park is in such a disturbed state due to past human impact.

Market valuation approaches, however, account only for small changes in the provision of commodities, assuming that marginal values increase when the flow of a certain product decreases, and the other way round (Turner et al., 2003). Such an approach does not take into account the distance to the threshold that can affect the value of ecosystem services (Limburg et al., 2002). Hence, it is rather a marginal change in the value of ecosystem resilience that should be in some way captured by markets (Brander et al., 2010). This is why study of the links between functional types of species and ecosystem processes deserves higher attention. The signal about the state of the functional types of species, if captured by markets, may serve to prevent large changes and the loss of other ecosystem values.

The results of this study suggest further research should give more attention to identifying ecosystem thresholds, since advanced inputs about conditions that can cause regime shifts, as well as about potential socio-economic implications, are of high importance for decision-making. Focusing on the links between functional types and ecosystem processes may bring insights that are more valuable for practical forest management and may serve well in creating incentives for the sustainable use of forest resources.

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REFERENCES

Angelstam, P., Roberge J-M., Dönz-Breuss M., Burfield I.J. and *G. Stähl* (2004). Monitoring forest biodiversity – from the policy level to the management unit. *Ecol. Bull.* **41**, 295-304.

Berger A.L. and *K.J. Puettmann* (2000). Overstory composition and stand structure influence herbaceous plant diversity in the mixed aspen forest of Northern Minnesota. *Am. Midl. Nat*. **143**(1), 111-125.

Brander, L., Gómez-Baggethun, E., Martin-López, B. and *M. Verma* (2010). The economics of valuing ecosystem services and biodiversity. In: *The Economics of Ecosystems and Biodiversity: The Ecological and Economic Foundations* (Ed. P. Kumar), 183-256. Earthscan, London.

Braun-Blanquet J. (1964). *Pflanzensoziologie – Grundzüge der vegetationskunde (Plant sociology – The study of plant communities)*. Springer Verlag, Vienna.

Chapin, F.S. III, Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L, Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. and *S. Díaz* (2000). Consequences of changing biodiversity. *Nature,* **405**, 234-242.

Contamin, R. and *A. Ellison* (2009). Indicators of regime shifts in ecological systems: What do we need to know and when do we need to know it? *Ecol. Appl.* **19**(3), 799-816.

Daily, G.C. (1997). *Nature's Services. Societal Dependence on Natural Ecosystems*. Island Press, Washington D.C.

Díaz S. and *M. Cabido* (2001). Vive la difference: plant functional diversity matters to ecosystem processes. *Trends Ecol. Evol.* **16**, 464-655.

Folke C., Holling, C.S. and *C. Perrings* (1996). Biological diversity, ecosystems, and the human scale. *Ecol. Appl.* **6** (4), 1018-1024.

Franklin, J.F. (1988). Structural and functional diversity in temperate forests. In: *Biodiversity* (Eds. E.O. Wilson and F.M. Peters), 166-175. National Academy Press, Washington D.C.

Franklin, J.F., Spies, T.A., van Pelt, R., Carey, A.B., Thornburgh, D.A., Berg, D.R., Lindenmeyer, D.B., Hamon, M.E., Keeton, W.S., Shaw, D.C., Bible, K. and *J. Chen* (2002). Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-Fir forests as an example. *Forest Ecol. Manag.* **155**, 399-423.

de Groot, R.S., Wilson M.A. and *R.M.J. Boumans* (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecol. Econ.* **41**, 393-408.

Holling, C.S. (1973). Resilience and stability of ecological systems. *Annu. Rev. Ecol. Syst*. **4**, 1-23.

Hooper, D.U., Chapin, F.S., Ewel, III, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M. Naeem, S., Schmid, B., Setälä, H., Symstad, A.J., Vandermeer, J., and *D.A. Wardle* (2005). Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecol. Monogr.* **75** (1), 3-35.

Hooper, D.U. and *P.M. Vitousek* (1997). The effects of plant composition and diversity on ecosystem processes. *Science*. **277**, 1302-1305.

Janković, M.M. and *V. Mišić* (1980). *Šumska vegetacija i fitocenoze Fruške gore.* Matica srpska, Posebna izdanja, Monografija Fruške gore, Novi Sad.

Lepš, J., Osbornová-Kosinová, J. and *M. Rejmánek* (1982). Community stability, complexity and species life history strategies. *Vegetatio*. **50**, 53-63.

Limburg, K.E., O'Neill, R.V., Costanza, R. and *S. Farber* (2002). Complex systems and valuation. *Ecol. Econ*. **41**, 409-420.

MA (Millennium Ecosystem Assessment) (2005). *Ecosystems and Human Well-being: Synthesis*. Island Press, Washington D.C.

O'Hara, K.L. (1998). Silviculture for structural diversity: a new look at multiaged systems. *J. Forest.* **96**, 4-10.

Magurran, A.E. (1988). *Ecological Diversity and its Measurement*. Croom Helm, London.

Pearce, D. (1998). Can non-market values save the tropical forests?, In: *Tropical Rain Forest: A Wider Perspective* (Ed. F.B. Goldsmith), 255-267. Chapman & Hall, London.

Perrings, C.A., Mäller, K.G., Folke, C., Holling C.S., and *B.O. Jansson* (1995). *Biodiversity Loss: Ecological and Economic Issues*. Cambridge University Press, Cambridge.

Posebne i opšta osnova gazdovanja šumama NP (2006-2015), (g.j. 3801-3814), 2006, Šumarski fakultet, Beograd.

Prostorni plan područja posebne namene Fruške gore do 2022. godine ("Službeni list AP Vojvodine", broj 18/04).

Putz, F., Redford, K., Robinson, J., Fimbel, R. and *G. Blate* (2000). *Biodiversity Conservation in the Context of Tropical Forest Management*. Biodiversity Series – Impact Studies, Paper no. 75, Washington DC: World Bank.

SCBD (Secretariat of the Convention on Biological Diversity) (2001). The value of forest Eco-systems. Montreal, SCBD, Technical Series 4.

Scheffer, M. and *S.R. Carpenter* (2003). Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends Ecol. Evol.* **18**(12), 648-656.

Scheffer, M., Carpenter S., Foley, J.A., Folke, C. and *B. Walker* (2001). Catastrophic shifts in ecosystems. *Nature*. **413**, 591-596.

Schwartz, M.W., Brigham, C.A., Hoeksema, J.D., Lyons, K.G., Mills, M.H. and *P.J. van Mantgem* (2000). Linking biodiversity to ecosystem function: implications for conservation ecology. *Oecol.* **122**, 297-305.

Shannon, C.E., and *W. Weaver* (1949). *The mathematical theory of communication*. University of Illinois Press, Urbana.

Spies, T.A. and *J.F. Franklin* (1991). The structure of natural young, mature, and old-growth Douglas-Fir forests in Oregon and Washington. In: *Wildlife and Vegetation of Unmanaged Douglas-Fir Forests* (Eds. L.F. Ruggiero, K.B. Aubry, A.B., Carey, M.H., Huff), 91-109. USDA, Forest Service, Portland, Oregon.

Staudhammer, L.C., and *V.M. LeMay* (2001). Introduction and evaluation of possible indices of stand structural diversity. *Can. J. For. Res.* **31**, 1105-1115.

Turner, R.K., Paavola, J., Cooper, P., Farber, S., Jessamy, V. and *S. Georgiou* (2003). Valuing nature: lessons learned and future research directions. *Ecol. Econ.* **46**, 493-510.

Uuttera J., Tokola, T. and *M. Maltamo* (2000). Differences in the structure of primary and managed forests in East Kalimantan, Indonesia. *Forest Ecol. Manag.* **129**, 63-74.

Walker, B.H, Abel, N., Anderies, J.M. and *P. Ryan* (2009). Resilience, adaptability, and transformability in the Goulburn-Broken catchment, Australia. *Ecol. Soc*. **14**(1), 12.

Walker, B.H., Gunderson, L.H., Kinzig, A.P., Folke, C., Carpenter, S.R. and *L. Schultz* (2006). A handful of heuristics and some propositions for understanding resilience in social-ecological systems. *Ecol. Soc.* **11**(1), 3.

Walker, B. and *J.A. Meyers* (2004). Tresholds in ecological and social-ecological systems: a developing database. *Ecol. Soc*. **9**(2), 3.

Wilson, E.O. (1992). *The Diversity of Life*. Norton, New York.

Zakon o šumama ("Službeni glasnik RS" broj 30/10).