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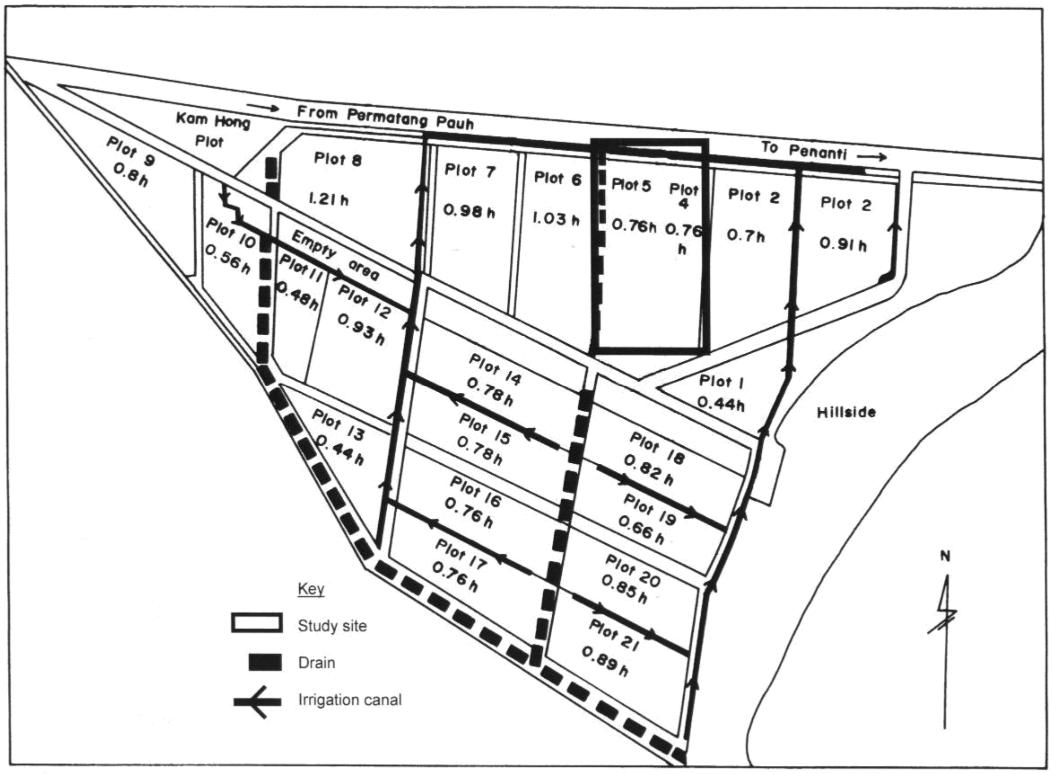

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RELATIONS OF PHYSICAL HABITAT TO FISH ASSEMBLAGES IN STREAMS OF WESTERN GHATS, INDIA

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Abstract. The present study, the influence of habitat structure on fish assemblages were assessed in fifteen selected streams of Western Ghats, India. Each stream 100m reach was quantified for depth, flow, velocity, fish cover, percentage of pool and riffle and fish density. Highest mean velocity (0.4 m/sec.) was recorded in Thalanai stream and deeper habitats were found in Kallar stream. High species diversity was found in Achankoil stream (H'=1.15) and low species diversity was recorded in Hanumannadhi stream (H' = 0.71). The physical habitat structure (depth, current and substrate) and cover complex were evaluated by using Evenness index (H'/H'_{max}). High diverse of physical habitat complex were encountered in Gugalthurai stream (E=2.8) and high value of cover complex was encountered in Sirkuli stream. Regression analysis showed that there was a significant correlation between habitat variables and fish abundance in all the sites and the cover complex values is not significantly correlated with abundance.

Key words: *fish assemblage, stream habitats, Western Ghats, India*

Introduction

Conservation of biodiversity requires an understanding of the processes involved in the structure and function of biotic communities. Western Ghats region of India is identified as one of the "hot spots" for biodiversity (Myers, 1990) and it is an important watershed of the Peninsular India. Water of mountain streams to lakes may look homogeneous but actually they are separated by variety of environmental factors such as temperature, depth, current and substrates into a great variety of habitats. The fauna of this habitat is known to have a very high degree of endemism. Although, the quality of habitat and variety of species have declined as a result of major changes in landscapes by human activities (Armantrout, 1995).

Resource management requires a better understanding of the condition of fish communities, their habitat requirements and the factors influencing them. In stream ecosystems, the diversity and community structure are influenced by water current, depth, substrates, nutrients and riparian cover, which determine the success or failure of community within the spatial distribution limits (Ricklefs, 1987). The influence of habitat structure and complexity on fish assemblage structure has been tested mostly in North American streams (Angermeier and Karr 1984; Capone and Kushlan,1991; Fausch and Bestgen, 1997; Gorman and Karr, 1978; Guisan and Zimmermann, 2000; Horig and Fausch, 2002; Oakes et. al., 2005; Schlosser, 1982, 1985, 1987) and

Australian streams (Bishop and Forbes, 1991; Pusey et al., 1993, 1995). Most of the information on fish habitat structure and assemblages are available from temperate streams and very meager information is available in Indian streams (Arunachalam et al., 1997; Arunachalam, 2000; Arunachalam et al., 2005). Hence, the present study addresses the influence of habitat structure on fish assemblage in fifteen different streams of Western Ghats.

Review of literature

Habitat structure has been identified as a major determinant in distribution and abundance of fishes from earlier time (Shelford, 1911). Later the zonation concept was developed by Huet (1954) where he explained the fish community in longitudinal succession with environmental characteristics. The environmental variation may have a significant impact on both assemblage structure and resource availability. Angermeier and Schlosser (1989) have examined the relative importance of habitat area, habitat volume, habitat heterogeneity and number of individuals as determinant of the species richness in a habitat patch. The influences of riparian vegetation (Cummins et al., 1989; Grefory et al., 1991; Ross, 1986), benthic organic matter (Cummins, 1974; Naiman and Sedell, 1979; Newbold et al., 1981a, b) in functional organization in stream community have been documented. Horwitz (1978) has proposed the stream order concept where the spatial heterogeneity associated with upstream versus downstream. The number of species increases in parallel with the stream order, which is attributable to an increase in habitat diversity and stability. Further more Vannote et al. (1980) have proposed the River Continuum Concept to explain the downhill movement of nutrients and organic matter from the riparian zone to the stream. With increase in stream order for each type of stream, the pressure on aboitic factors gradually decreases as spatial heterogeneity and stability improve. Moreover, human impact had now become a factor which modifies the spatial structure of fish community, for example marked changes in flow regime and the water quality (Bovee, 1982). The baseline study on the assemblage structure of fishes in south Indian streams was addressed by Arunachalam (2000) and similar work on Sri Lankan streams also available (De Silva et al., 1980; Kortmulder, 1987; Kortmulder et al., 1990).

Study Area

Western Ghats is a chain of hills of 1600 km in length running parallel to west-coast of Peninsular India (between 8º and 21º N latitudes) from the mouth of river Tapti in Gujarat to Kanyakumari in Tamil Nadu. The four major rivers flowing in the east are Godavari, Krishna, Cauvery and Tamiraparani, while the river Bharathapuzha, Periyar and Chaliyar in Kerala flow towards the west. There are number of numerous quick flowing streams and rivers arising on the western slope discharging into the Arabian Sea. In the present study fifteen streams covering major river basins representing from Tamil Nadu, Kerala and Karnataka states are selected. Summary of the study streams and their general features are given in *Table 1*.

Table 1. *Summary of study sites general features in the Western Ghats, India.*

Sites	River basin East/West flowing	Latitude/ longitude	Altitude (m)	Stream order	Stream gradient (%)	Air temp (°C)	Mean width (m)
S1-Samikuchi	Chittar - II West flowing	8° 25' N 77° 25' E	500	3	7	28	45.6
S2-Thalayanai	Manimuthar East flowing	8° 35' N 77° 25' E	300	3	7	32	25.6
S3-Karaiyar	Tamiraparani East flowing	8° 40' N 77° 20' E	300	3	7	30	13.8
S4-Hanumannadhi	Chittar East flowing	9° 05' N 77° 20' E	200	3	2	30	37.4
S5-Gugalthurai	Cauveri East flowing	11° 40' N 76° 45' E	600	3	6	30	11.5
S6-Kallar	Vamanapuram West flowing	8° 45' N 77° 15' E	800	3	7	28	22
S7-Achankoil	Achankoil West flowing	9° 10' N 76° 50' E	600	4	4	27	20
S8-Panniyar	Periyar West flowing	9° 45' N 77° 15' E	912	3	6	28	14
S9-Thalipuzha	Cauvery East flowing	11° 30' N 76° 15' E	750	3	3	31	9.1
S10-Bavalipuzha	Cauvery East flowing	11° 55' N 76° 45' E	1350	3	4	31	20
S11-Ekachi	Cauvery East flowing	12° 45' N 75° 45' E	700	3	4	26	21
S12- Kigga	Thunga East flowing	13° 20' N 75° 15' E	900	3	8	20	9.5
S13-Thunga	Thungabadhra East flowing	13° 45' N 76° 20' E	600	5	1	26	80
S14-Sirkuli	Aghanasini West flowing	14° 30' N 74° 45' E	900	3	4	34	55
S15-Ganeshpal	Bedti river West flowing	14° 15' N 74° 45' E	700	4	3	29	75

Materials and methods

Quantification of habitat characteristics and habitat inventory were followed by the methods described in Arunachalam (2000). Inventory was carried out at a fixed point, which is designed as a reference point. Each stream a 100 m reach was quantified for depth, flow and substrate characteristics. Number of transects usually 5-10 were taken across the stream channel, the depth, water velocity and dominant substrates were measured or estimated at 0.5 or 1 m intervals across the transects. Water velocity was recorded with a digital electronic Pigmy water current meter (Model: Propeller type no. Lynx pp. 001). The depth measurement were used to determine the proportion of the habitat within six depth categories (D1–6) corresponding to the 0-10, 11-30, 31-60, 61-100, 101-150 and >150cm, respectively. Water velocity was grouped into four categories (F1-4): zero, low, moderate and fast corresponding to 0-0.15, 0.16-0.30, 0.31-0.60 and >0.60m sec^{-1} respectively. Substrate was classified as Bedrock (>512mm diameter), boulder (128-512mm), cobble (64-128), gravel (16-64mm), sand (1-16) and leaf litters. Fish cover was classified into seven categories: No cover, Small boulder undercut, Boulder undercut, Submerged log, Overhanging vegetation, Bedrock undercut and Root undercut. The number of unique configuration of each category and their frequencies of occurrence were used to compute Evenness index (H'/H'_{max}) for each parameters. These index values for depth, current and substrate were summed to give overall measures of physical habitat complexity with a maximum value of three. A total habitat complexity index (Physical + cover) was then estimated by summation of the physical and cover components (Pusey et al., 1995). Area (length x mean width of the channel), Volume (area x mean depth) and % of Pool-riffle habitat in 100m reach of each site were estimated based on Angermeier and Schlosser (1986). Riparian cover in the site was estimated using spherical Densiometer (model: C).

Fish sampling was performed in individual habitats using mono-filamentous gill nets (mesh size 8 to 25 mm), cast net and dragnets. Based on the fish catch and underwater observation, species richness (S) and fish abundance data were generated for each site Pusey et al. (1995) Relationship among number of individuals, habitat areas, habitat volume, % of pool and riffle, % of riparian cover and habitat complexity were examined using linear regression (Angermeier and Schlosser, 1986). Except habitat complexity all other data were $\log_{10}$ – transformed in the analysis in order to minimize effects of non-normality.

Results

Structural characteristics such as mean channel width, mean depth and mean flow were generally varied among the study streams. *Table 2* shows the major structural features of the study sites. Deeper habitats were found in Kallar stream (mean depth 98.6 cm). Mean stream width was varied from 9.1 to 80 m among study stream. Highest mean velocity (0.4 m/sec.) was recorded in Thalayanai stream of Tamilnadu region. *Table 3* shows the major physical habitat variables and the biotic variables. Among the fifteen streams, the high species diversity was found in Achankoil stream (H' = 1.15), next to that the Kallar stream (H' = 1.14) had greatest species diversity and low species diversity was recorded in Hanumannadhi stream (H' = 0.71). Physical habitat complexity index (Physical + cover) ranged from 2.22 to 2.83. Highest habitat complexity was recorded in Gugalthurai stream whereas in Sirkuli stream the cover complexity was high (*Fig. 1*). Habitat volume was high in Thalaiyanai stream, which inhabits greater density of fishes (595 in 100 m reach).

Table 4 shows the result of regression analysis between habitat characteristics and fish abundance. There was a positive correlation between habitat characteristics and fish abundance in all the sites and the results were highly significant ($p > 0.01$) (Habitat volume $r^2 = 0.53$; Habitat area $r^2 = 0.66$; Physical habitat complex $r^2 = 0.76$), whereas in the cover complex values is not significantly correlated with abundance. Regression analysis also showed that habitat complexity, habitat volume, habitat area, instream cover and percentage of pools-riffles had some capability of predicting fish abundance.

Table 2. *Structural characteristics of study streams of Western Ghats, India.*

Sites	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**
Mean width (m)	45.6	25.6	13.8	37.4	11.5	22	20
Mean depth (cm)	53.2	84.3	51.8	23.3	53.5	98.6	34.8
Depth (%)							
Depth 1	13.0	4.7	6.9	4.3	0	6	2
Depth 2	30.4	9.5	37.9	56.6	31.8	15	23
Depth 3	19.6	26.5	20.7	34.8	27.3	30	42.7
Depth 4	29.6	31.0	20.7	4.3	31.8	24.5	26
Depth 5	7.4	14.3	10.3	0	9.0	20.5	0
Depth 6	0	14.3	3.5	0	0	4	6.3
Mean flow (V = m/sec)	0.23	0.4	0.27	0.17	0.19	0.28	0.32
Flow (%)							
Stagnant	37.5	25	0	55.5	46.0	15	33
Slow	25.0	0	80	44.5	23.3	42.2	15.5
Moderate	37.5	50	20	0	30.7	25	36.5
Turbulent	0	25	0	0	0	17.5	15
Substrates (%)							
Bedrock	57.5	30.8	11.2	26.5	14	9.5	9
Boulder	18.2	40.0	33.7	18.6	24	12.5	12
Cobble	12.3	12.4	15.0	26.5	30	22.0	22
Gravel	3.0	10.0	3.8	0	8	21.5	21
Sand	7.5	4.8	32.5	28.4	17	26.5	28
Leaf litter	1.5	2.0	3.8	0	7	8.0	8
Fish covers (%)							
No cover	0	0	10	10	0	0	15
Small boulder undercut	31	18	22	22	13	37	0
Boulder undercut	28	24	17	17	30	30	0
Submerged log	7	6	6	6	0	0	15
Overhanging vegetation	3	13	17	17	18	15	46
Bedrock undercut	31	24	24	0	30	12	0
Root undercut	0	15	15	17	9	6	24

Sites	**S8**	**S9**	**S10**	**S11**	**S12**	**S13**	**S14**	**S15**
Mean width (m)	14	9.1	20	21	9.5	80	55	75
Mean depth (cm)	10	38.9	69.0	36.3	54.3	84.7	52.9	62.5
Depth (%)								
Depth 1	10	3	0	0	0	0	17.2	0
Depth 2	33	39.4	11.1	35	21.4	21.1	38.0	18.2
Depth 3	37	36.4	28.9	65	46.4	26.3	13.8	27.2
Depth 4	20	21.2	51.1	0	28.6	26.3	17.2	36.4
Depth 5	0	0	8.9	0	3.6	7.9	3.5	18.2
Depth 6	0	0	0	0	0	18.4	10.3	0
Mean flow (V = m/sec)	0.3	0.20	0.14	0.2	0.3	0.27	0.27	0.06
Flow (%)								
Stagnant	70	50	60	46.2	50.0	11.1	38.5	90
Slow	20	33.3	10	23.0	16.7	44.4	23.0	10
Moderate	0	16.7	30	30.8	33.3	44.4	30.8	0
Turbulent	0	0	0	0	0	0	7.7	0
Substrates (%)								
Bedrock	23	8	47	35	70.8	10	19.5	60
Boulder	20	42	29.4	13.3	10.8	-	9.7	20
Cobble	14	6	2.0	21	6.6	-	25.3	-
Gravel	12	3	1.9	9	1.4	10	13.4	10
Sand	20	31	14.7	16.8	6.6	80	17.9	10
Leaf litter	11	10	5.0	4.9	3.8	-	14.2	-
Fish covers (%)								
No cover	0	18	10	30	0	67	13	25
Small boulder undercut	40	27	20	26	23	0	17	10
Boulder undercut	12	27	40	17	12	22	13	25
Submerged log	0	0	0	0	0	0	0	0
Overhanging vegetation	20	14	10	9	32	0	13	0
Bedrock undercut	16	9	10	9	12	11	22	35
Root undercut	12	5	10	17	5	0	22	5

Table 3. *Physical habitat variables and biotic variables in the study streams of Western Ghats, India.*

Streams	Species richness (S)	Species diversity (H')	Habitat		In-stream cover (%)	Pool, riffles (%)	Habitat complexity		Fish density (Nos.)
			area (m^2)	volume (m^3)			Physical	Cover	
Samikuchi	12	1.03	953	306.73	50	71	2.655	0.854	354
Thalayanai	18	1.11	1103	628	80	63	2.664	0.956	595
Karaiyar	10	0.94	884	326.62	75	80	2.433	0.962	278
Hanumannadhi	6	0.71	589	206.4	20	60	2.675	0.961	239
Gugalthurai	15	1.08	912	428.23	60	65	2.833	0.94	407
Kallar	17	1.14	924	265.5	75	63	2.788	0.893	521
Achankoil	17	1.15	964	288.54	80	69	2.696	0.915	568
Panniyar	10	0.90	597.8	225.34	70	53	2.671	0.926	249
Thalipuzha	9	0.88	764.1	367.63	25	65	2.564	0.925	239
Bavalipuzha	15	1.08	842	267.69	65	63	2.381	0.898	296
Ekachi	11	1.01	659	233.5	65	79	2.805	0.946	306
Kigga	11	0.97	399	147.11	85	65	2.335	0.899	246
Thunga	16	1.16	896	525.8	40	65	2.419	0.768	398
Sirkuli	11	0.95	1115	388.47	65	71	2.78	0.984	307
Ganeshpal	9	0.91	755	327.42	30	66	2.222	0.895	281

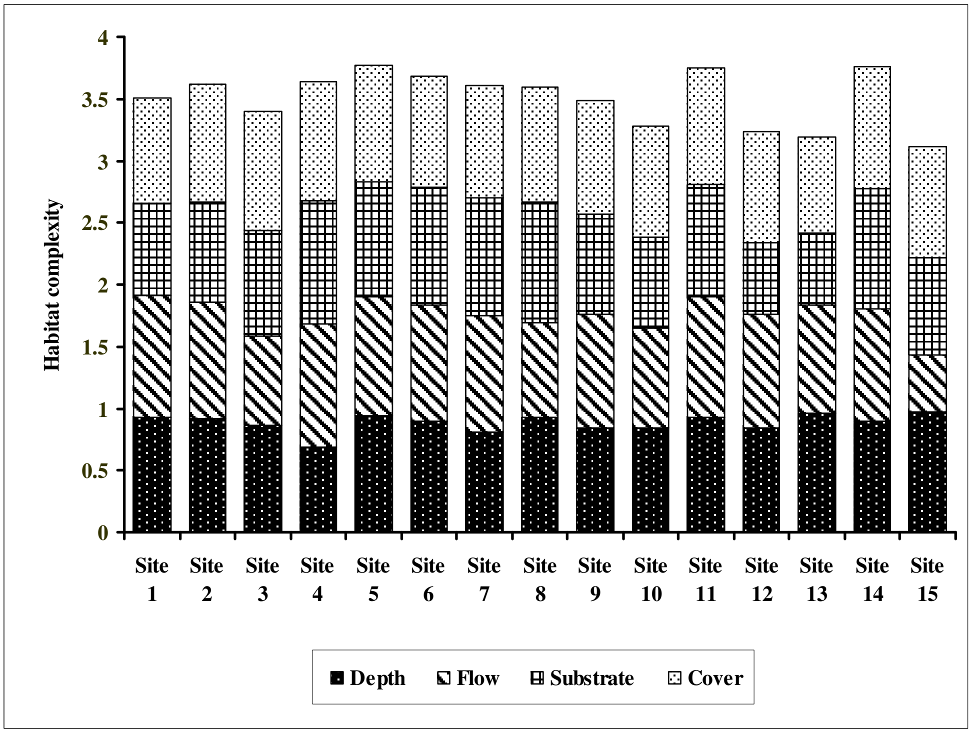

Figure 1. *Habitat complexity index of fifteen streams of Western Ghats, India.*

Table 4. Regression of fish abundance vs. habitat area, habitat volume, instream cover, percentage of pool-riffle and habitat comlexity.

Variables	Intercept B	Slope A	r^2
Habitat area	0.58	1.45	0.66*
Habitat volume	0.62	0.91	0.53*
Instream cover	0.66	0.08	0.45
% of pools-riffles	0.21	1.31	0.56*
Physical Habitat complex	1.11	-1.23	0.76*
Cover complex	0.16	2.90	0.32

*P<0.01

Discussion

The physical habitat (depth, current and substratum) forms the ‘structure’ within which an organism makes its home. This habitat structure determines the abundance and diversity of organism (Baretto and Uieda, 1998; Hubert and Rahel, 1989; Hynes, 1970; Pusey et al., 1993; Schlosser, 1982). The basic pattern of increasing species richness and low replacement are consistent with the hypothesis based on habitat diversity (Horwitz, 1978). Importance of habitat structure has been identified as the primary basis on which many biological communities are organized (Schoener, 1974) and several studies have supported this generalization for fish communities (Aadland, 1993; Angermeier and Karr, 1984; Angermeier and Schlosser, 1986; Bain et al., 1988; Evans and Noble, 1979, Jackson et al., 2001, Lohr and Fausch, 1997, Matthews et al., 1994, Pusey et al., 1995; Romanuk et al., 2006; Schlosser and Toth, 1984; Schlosser, 1982; Tallman and Gee, 1982). The organization of fish assemblages in the present study also follows the uniform pattern reported from other regions. Also Williams (1964) emphasized that in larger surface areas there will be many habitats and the fauna will increase when the surface area increases. In the present study significant correlation between fish species abundance and habitat area supports the hypothesis. However, in aquatic environment, the third spatial dimension (i.e., depth) can be included in habitat patch (Angermeier and Schlosser, 1986). The volume predicted fish abundance more than habitat area, thereby suggesting that the area and depth of stream habitat also influence distribution of stream fishes (Angermeier and Schlosser, 1986, Harvey and Stewart, 1991; Pusey et al., 1995). The influence of depth on fish abundance in the present study also falls in line with the earlier findings.

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THE INFLUENCE OF ROUTINE AGRICULTURAL ACTIVITIES ON THE QUALITY OF WATER IN A TROPICAL RICE FIELD ECOSYSTEM

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Abstract. Physico-chemical parameters of the rice field water of Bukit Merah Agricultural Experimental Station (BMAES), Penang, Malaysia were investigated during two cultivation seasons of 2004/2005. The selected variables showed wide ranges of variation. Except for total organic matter (TOM) and total suspended solid (TSS), the values of all variables were significantly different at all sampling occasions. Most of the physico-chemical parameters showed significant relationship to the water level either in the presence of rice plant or after harvesting of the crop. The rice agroecosystem is a complex ecosystem due to the interaction of aquatic organisms, including algae and macrophytes. This unique swampy ecosystem is usually disrupted during ploughing, drainage, harvesting and application of fertilizers and pesticides. It was concluded that the agricultural activities in this rice field contributed effectively in creating the wide variations in the values of physico-chemical parameters.

Keywords: *water parameters, rice field ecosystem, agricultural activities*

Introduction

The rice field is a unique man-made environment with high diversity of aquatic organisms. In South East Asia rice fields cover over 70 million hectare (Lim, 1990). The rice fields have been recognized as an important source for producing cheap protein, such as fish and crab (Ali and Ahmad, 1988) and form an important component in the economic life of the rural people (Lim et al., 1984). The rice field ecosystem is a dynamic and highly disturbed ecosystem. Routine agricultural practices including ploughing, draining, fertilizer and pesticide applications, harvesting and desiccation brought about the wet and dry climate cycles influenced the diversity the aquatic community (Che Salmah et al., 1998). Generally, rice fields show a wide variation of water parameters such as temperature, pH, dissolved oxygen, conductivity, nitrate and phosphate (Lim, 1990). This study was aimed at investigating the effect of rice cultivation practices in the rice agroecosystem on the quality of rice field water.

Materials and methods

Description of the study area

This study was conducted at the Bukit Merah Agricultural Experimental Station (BMAES), in Permatang Pauh, Seberang Perai, Penang. The station is located in Bukit Merah, about 20 km from Butterworth at latitude 5.417°N and longitude 100.417°E. The 16.5 ha irrigated research station is administered by the Agriculture Centre of Bukit Merah, Seberang Perai under the supervision of Malaysian Agricultural Research and

Development Institute (MARDI) of Bumbung Lima, Seberang Perai. Rice seeds (variety MR220) are produced at this station and distributed to the farmers in surrounding areas (*Fig. 1*).

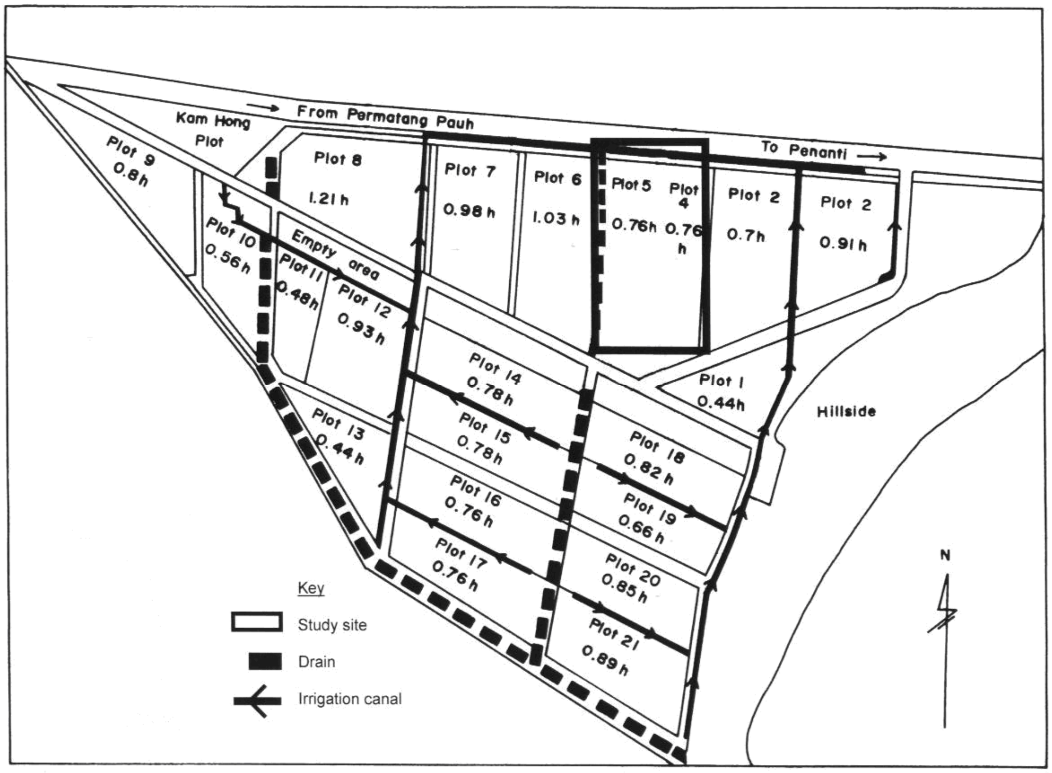

Figure 1. *Layout plan of study area at Bukit Merah Agricultural Experimental Station*

Environmental study

The water layer closer to the sediment (approximately 15 cm deep), was easily measured *in situ* for their pH, electrical conductivity (EC), temperature and dissolved oxygen (DO) contents. Dissolved oxygen (mg L^{-1}) and water temperature (°C) were measured with a DO meter (YSI-57), whereas the pH of the water was recorded using a Termo Orion-Model 210 pH meter. The water conductivity (µS cm^{-1}) was measured with an SCT meter (YSI-55). Water samples were collected from the sampling area in 500 ml plastic bottles. These bottles were sterilized with concentrated HCl (Ali *et al.*, 2002) prior to sampling and rinsed three times with rice field water before taking the actual water samples. They were kept in a cool Coleman® chest during transportation to the laboratory. In the laboratory, the samples were transferred into a freezer (-20°C) prior to analysis (Che Salmah et al., 1998). The measurements of each variable were replicated three times and the mean and range values were recorded.

The Total Organic Matter (TOM) of the sediment was analysed following the method of (Faithful, 2002). First, the soil was air dried at room temperature. Then it was crushed in a mortar and pestle and sieved using a 0.5 mm sieve. Ten gm of the sieved soil (W_0) was weighed using a digital balance (A&D Model ER-180A, Japan) and placed in a dry and clean crucible disc. It was dried in an oven (Mermmert model 854

Schwabach, Germany) at 100 °C overnight. The dry sample was recorded as W_1. The weight of the sample after water elimination was calculated as = W_0 - W_1 = W_2. The sample was placed in a furnace (Blue M Model M15A-1A) at temperatures of 450-500 °C for 3-4 hours. The sample was then reweighed and labelled as W_3. The total organic matter (TOM) was calculated as TOM % = W_3 / W_2 X 100. These procedures were replicated three times and the mean values were recorded.

Total Suspended Solids was determined using the method of (Tomar, 1999). A 500 µm filter paper was dried overnight in an oven (Mermmert model 854 Schwabach, Germany) at 100 °C. It was weighed using a digital balance (A&D Model ER-180A, Japan) and labelled as F_1. The filter paper was placed in the filtration funnel (Millipore®, USA) and 100 ml of water sample was poured into it. When all water has passed through the funnel, the filter paper with filtered material was removed from the funnel and dried overnight in an oven at a temperature of 100 °C. It was reweighed and labelled as F_2. The total suspended solids were calculated as F_2 - F_1. Three samples were analysed and the mean values were recorded.

The nitrogen-nitrate and phosphate contents in the water were determined using a YSI 9100 Photometer test kit. A water sample was filled in a test tube to 10 ml mark and the appropriate reagents for the tests were added following the kit procedures. The mixture was left 10 minutes before taking the light absorbance readings at selected wavelength. The wavelength set for the phosphate content was 640 nm and the nitrate content was read at 570 nm. Three readings were taken for each compound and the mean values were recorded.

Results

Selected physico-chemical variables of the rice field showed wide ranges of variations (*Table 1*). Water level varied widely depending on rice cultivation stages. Water was either drained or coincidentally low due to the absence of rain during plough, seeding, harvest up to the early part of the fallow phases. Growth patterns in the field indicated there were two rice crops; from late September 2004 to late of December 2004 and from early April 2005 to late July 2005. These rice crops were separated by growth of grasses and other weeds during fallow phase.

The pH however, showed smaller ranges with a mean of 6.27 rendering the field a slightly acidic environment. In general the conductivity of the water in the study area was quite high and ranged between 15 and 250 $\mu S\ cm^{-1}$. Biweekly mean nitrate were relatively high in this study area and the highest concentration (2.71 $mg\ L^{-1}$) was detected in late August 2004. The mean of soluble phosphate levels were relatively high and ranged from 0.02 to 1.83 $mg\ L^{-1}$. The amount of total organic matter (TOM) in the sediment varied according to the physical and chemical conditions of the field. The mean of TOM in the study area was considered high (5.68%). Except for total organic matter (TOM) and total suspended solid (TSS), the values of all variables were significantly different at all sampling occasions (Kruskal-Wallis at $P < 0.05$).

Table 2 shows the relationship of these variables to the fluctuation of water level. The pH, dissolved oxygen and total organic matter (TOM) showed no relation to the water level in the field. However, other variables showed a significant correlation with the water level (*Fig. 2: a-e*). The water was considered as a key factor controlling other variables and the changes in other variables values were closely followed the fluctuation in water level.

Table 1. *Mean and ranges of water parameters, rice plant height and total organic matter in the sediment of rice field plots.*

Parameter	Mean ± SE	Range
Water Level (cm)	13.99 ± 0.486	1.5 – 35
Plant Height (cm)	51.69 ± 1.893	0 – 130
pH	6.271 ± 0.021	5.15 – 7.7
Conductivity (µScm^{-1})	84.06 ± 2.283	15 – 250
Dissolved Oxygen (mg l^{-1})	2.87± 0.073	1.06 – 6.87
Temperature (°C)	29.21 ± 0.010	23 – 34
Total Organic Matter (%)	5.68 ± 0.062	3.03 – 8.77
Total Suspended Solids (mg l^{-1})	48 ± 0.11	5 –110
Phosphate (mg l^{-1})	0.43 ± 0.020	0.02 – 1.83
Nitrate-Nitrogen (mg l^{-1})	0.57 ± 0 .030	0.044 – 2.713

Table 2. *Correlation coefficient of physico-chemical variables related to the water level in the rice field.*

	Correlation Coefficient	Significance
Dissolved Oxygen	0.004	NS
pH	0.03	NS
Conductivity	-0.271	**
Temperature	-0.235	**
Total Organic Matter	0.029	NS
Total Suspended Solids	-0.155	*
Phosphate	-0.329	**
Nitrate Nitrogen	-0.409	**

* Significance level at $P < 0.05$
** Significance level at $P < 0.01$, NS = Not Significant

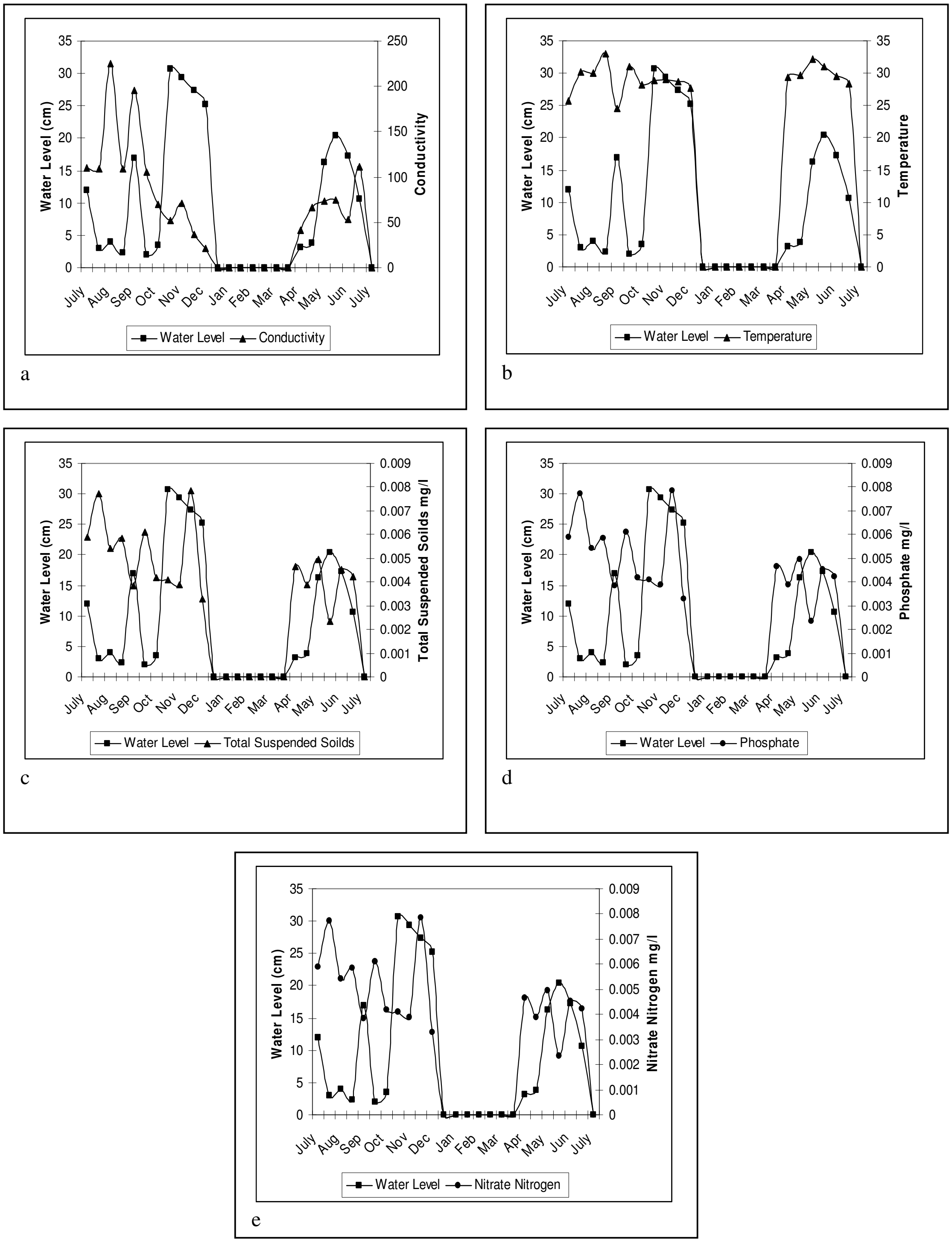

Figure 2. *Water level in the rice field in relation with a: conductivity, b: temperature, c: total suspended solids, d: phosphate, e: nitrate nitrogen.*

Discussion

Generally, this field showed wide ranges in physico-chemical conditions such as temperature, dissolved oxygen, pH, and conductivity due to its shallowness, growth of macrophytes and inputs of nutrients (Lim, 1990).

Ali and Ahmad (1988) reported that the water temperature in the tropics varies from as low 24 ºC in the morning or during cloudy or rainy days and up to 40 ºC on hot sunny days. The temperature of the water in the field is affected directly by the height of the plants. In this rice field the water temperature was varied from 23 to 34 with a mean of 29.21. These values fell with the reported temperature range but slightly lower possibly due to difference in geographical locations and irrigation regime as well. In a rain fed rice field, Che Salmah et al. (1998) reported that the water temperature of a rain fed rice field ranged from 26.4 to 39.7 ºC. Lim et al. (1984) recorded that the mean temperature in a rice field could reach as high as 40.1 ºC due to shallow water. When exposed to strong sunlight, the water was heated up readily, especially in the early stages of the rice growing phases when it lacked vegetational cover.

In the rice field the amount of dissolved oxygen increases due to photosynthesis by algal populations. In the present study, the dissolved oxygen was fairly low with a mean of 2.87 mg L^{-1}. In a rain fed rice field, Che Salmah et al. (1998) recorded that dissolved oxygen levels fluctuated considerably, ranging between 1.65 to 5.68 mg L^{-1}. it has been reported that the strong wind passing and shallow water of the rice fields create greater water turbulence, which is rich in dissolve oxygen (6.7 mg L^{-1}). In contrast, stagnant water at the edge of rice field has very little oxygen (1.7 mg L^{-1}) (Ali and Ahmad, 1988). However, Lim et al. (1984) recorded that the dissolved oxygen of another Malaysian rice field in Tanjung Karang was very high with a mean 8.6 mg L^{-1}. Similarly, in one of the Netherlands rice field, Fores (1992) recorded a wide range of dissolved oxygen from 2.83 to 11.1 mg L^{-1}.

Generally, the pH values in paddy fields differ following rice phases during the cultivation period (Fores, 1992). In this study, the pH mean showed that the rice field was slightly acidic and the value similar to what have been reported in other Malaysian rice fields earlier (Lim et al., 1984 and Che Salmah et al., 1998). The decomposed rice stalks from previous season and weeds especially after herbicide applications, contribute to the acidity of the rice environment (Ali and Ahmad, 1988). Additionally, pH is influenced by the metabolic activity of the flora particularly photosynthesis. The lower values of the pH could also be attributed to the presence of substantial amounts of organic matter, associated with CO_2 production and nitrogen mineralization (Ali and Ahmad, 1988).

The values of recorded conductivity was lower compared to those reported in a Malaysian rice field study conducted by Lim et al. (1984) but it was higher compared to those in the study of Che Salmah et al. (1998). However, they were similar to the mean values in the study of Helliwell and Stevens (2000) who recorded the same mean values of conductivity in a rice field in New South Wales in Australia of 90.2 and 172.3 µS cm^{-1}. These differences of conductivity values were due to the inflow of water, nutrients cycling within the rice fields and fertilization process (Helliwell and Stevens, 2000; Stevens et al., 2006). The applications of fertilizers and pesticide contribute significantly in changing the ion contents of the water and the chemical properties as well (Lim, 1990).

Seasonal factors could have accounted for the fluctuations in NO_3 and NH_4 levels. Rainfall which supplies mineral nitrogen in the form of NO_3 and NH_4 is an important factor (Heckman, 1979). Other probable causes of NH_3 fluctuations were fertilizer applications and high utilization rate by rice plants (Ali and Ahmad, 1988). The concentration of nitrate-nitrogen was higher than that observed in the rice fields in Titi Serong (Ali and Ahmad, 1988) which ranged from 0.22 to 0.75 mg L^{-1}. The difference was justified as the effects of the fertilization process (Ali and Ahmad, 1988), agricultural activities and high activity of the nitrification bacteria (Law and Mohammad Mohsin, 1980). While the mean value of phosphate in the rice field water (mainly due to the fertilization and pesticide application), was high compared to those observed in streams (Ali et al., 1998, 2002, 2003).

The nature of the muddy environment of rice agroecosystem indicates high values of total suspended solids. Although the high densities of phytoplankton in the rice field contributed to increase of contents of suspended solids (Che Salmah, 1996), the mean value of total suspended solids in this rice field was lower than those observed in the polluted Kelang River (Law and Mohammad Mohsin, 1980). It is quite obvious that there is a great difference between lotic and lentic environments, that the streams receive higher level of disturbance and turbidity due to the running velocity. However, high value of total organic matter of this rice field sediment was the result of fertilization during cultivation activities that supported dense floral growth and decaying of macrophytes.

Usually phosphate concentrations are high during the early growing season due to the fertilization process in the field. Low concentrations appear at the end of the season when most of the phosphate has been utilised by rice plants (Cheah and Lai, 1990). Fertilizations and insecticide application might have caused increases of phosphate concentration in the study field. High levels of phosphate concentration could be related to low water levels which might cause muddy environment (Che Salmah, 1996) as well as the phosphate utilisation rate of rice plants (Cheah and Lai, 1990).

Like other aquatic ecosystem, the nutrient cycle and ecological interaction between the aquatic communities in the rice field could cause intermittent changes in chemical and physical characteristics of the rice field water. Additionally, in the rice field the cultivation practices such as plouging, harvesting, desiccation, and application of chemicals, appeared to have a significant role in changing the quality of the field water physically and chemically.

Conclusion

The rice field is a unique man-made environment with variety of aquatic organisms and their dynamic population, abundance and diversity is closely related to the environmental changes in the rice fields. This study investigated the effect of the cultivation practices on the physical and chemical properties of the water. These findings will effectively contribute toward more understanding about the chemical and physical environment of the rice agroecosystem.

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ALGAL INDICATION OF POLLUTION IN THE LOWER JORDAN RIVER, ISRAEL

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Abstract. The Lower Jordan River is one of the most polluted in Israel. We used algal bio-indicators to monitor pollution levels in the dry and rainy seasons in order to assess a self-purification capacity of the aquatic ecosystem. We used 135 species of algae and cyanobacteria as indicators of pH, salinity, and organic pollution. The indices of saprobity (S and EPI), river pollution, RPI, and toxicity, BI testify to a greater anthropogenic impact in the lower section, partly owing to influx of pollutants with rain water during the wet season. The aquatic ecosystem state index (WESI) indicates low self-purification activity of biotic communities in this part of the river flow. Multiple Regression Analysis highlighted salinity and trophic elements as the major impact factors for algal diversity. A combination of bio-indication metod, including new integral indices and statistic analysis, and bioassey were found most efficient for monitoring of river ecosystems in Israel.

Keywords: *Lower Jordan River, water quality, algal bio-indication, season, Israel*

Introduction

During the last millennium, aridity increases in the Mediterranean Region leading to a loss of water resources, which is aggravated by the ever increasing anthropogenic impact (Körner et al., 2005). At the some time, this threatening situation places high demands on monitoring of water resources. However, in such countries as Israel, the hydrobiological indicators were scarcely used so far (Israel Ministry of the Environment, 2005). Yet self-purification of water bodies depends on aquatic biota and mainly on algae as the primary producers. Therefore, algal diversity is indicative of the ecological and sanitary status of the river as well as the self-purification potentials (Janauer and Dokulil, 2006). Our approach is based on bio-indication of algal habitats, including not only the diversity, but also the density of each indicator species. The assessments of aquatic ecosystems based on algological indicators are widely used (Whitton and Rott, 1995; WFD, 2000; Barinova et al., 2006a), but their potentials are not as yet fully recognized. In particular, mapping of water quality domains based on the saprobity indices, obtained from the analysis of algal communities (Sumita, 1986; Watanabe et al., 1986), has been efficient in our studies of the Upper Jordan River (Lipkovsky et al., 2007) and can be applied to the Lower Jordan River as well.

Algae have long been identified as valuable indicators in the bio-monitoring of stream and river ecosystems (Kolkwitz and Marsson, 1908 altered by Hill et al., 2000). More recently, bio-monitoring has been applied to a variety of water quality problems (Hill et al., 2000; Potapova and Charles, 2003). Algal communities provide an integrated measure of water quality as experienced by the aquatic biota and have many biological attributes that make them ideal for biological monitoring. About 100 algal species were founding in the Lower Jordan River (Ehrlich, 1995; Lange-Bertalot in Nevo and Wasser, 2000; Rayss, 1951), but their potentials for algal bio-monitoring were not realized so far.

Study area

The Jordan River is the largest river in Israel, divided into the upper and lower reaches, with the Lake Kinneret in between. The Upper Jordan River discharges into the Lake Kinneret. The outlet from the lake is blocked by the Alumot-dam, which has not been overflowed since 1991 (Barel-Cohen et al., 2006). The Lower Jordan River extends between the Alumot-dam (approximately 210 m below sea level) and the Dead Sea (approximately 417 m below sea level) with a catchment area of about 15,000 km^2 (Salameh 1996). The river is about 105 km long with approximately 190 km meandering distance (Hamberg, 2000). The characteristics of the Lower Jordan River are much different than those of the Upper Jordan River. While the Upper Jordan River is a source of drinking water, the major water sources of the Lower Jordan River are the Saline Water Carrier and the effluents from the Bitania wastewater treatment plant. The Saline Water Carrier contains a mixture of saline spring water diverted from the western shore of the Lake Kinneret and the treated urban sewage effluents. The Bitania source consists of poorly treated waste effluents.

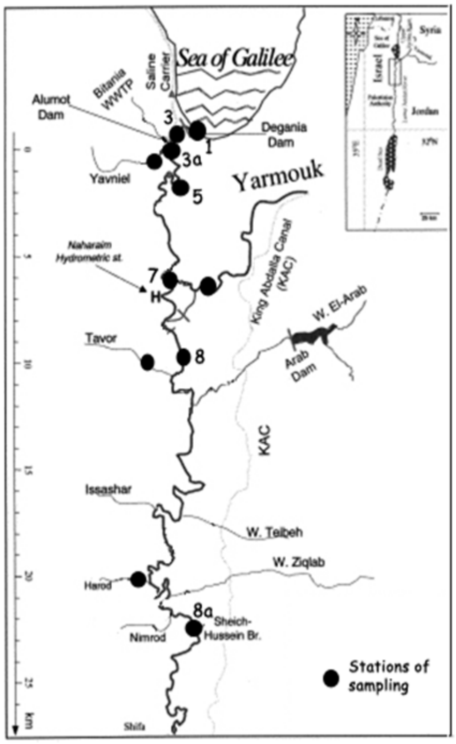

Figure 1. *Map of study site*

For the purposes of this study, the taxonomic composition of algal assemblages, species densities, and indicator species were determined for the northern part of the Lower Jordan River (*Figure 1*) between the outlet from the Lake Kinneret (station 1) and the Hussein Bridge station (station 8a), as well as for its tributaries (Yavniel, Yarmuk, Harod Tavor, and Bezet Rivers) (*Figure 2*). Both, the major canal and the tributaries are heavily polluted. The study was conducted during the dry and wet seasons of the summer-dry climate of Israel. The indices of saprobity, water pollution, and self-purification capacities were calculated for all the sampling sites (Sumita, 1986). The algal assemblages of the Lower Jordan River summarily reflect the ecological status and dynamics of the unique river flowing below sea level.

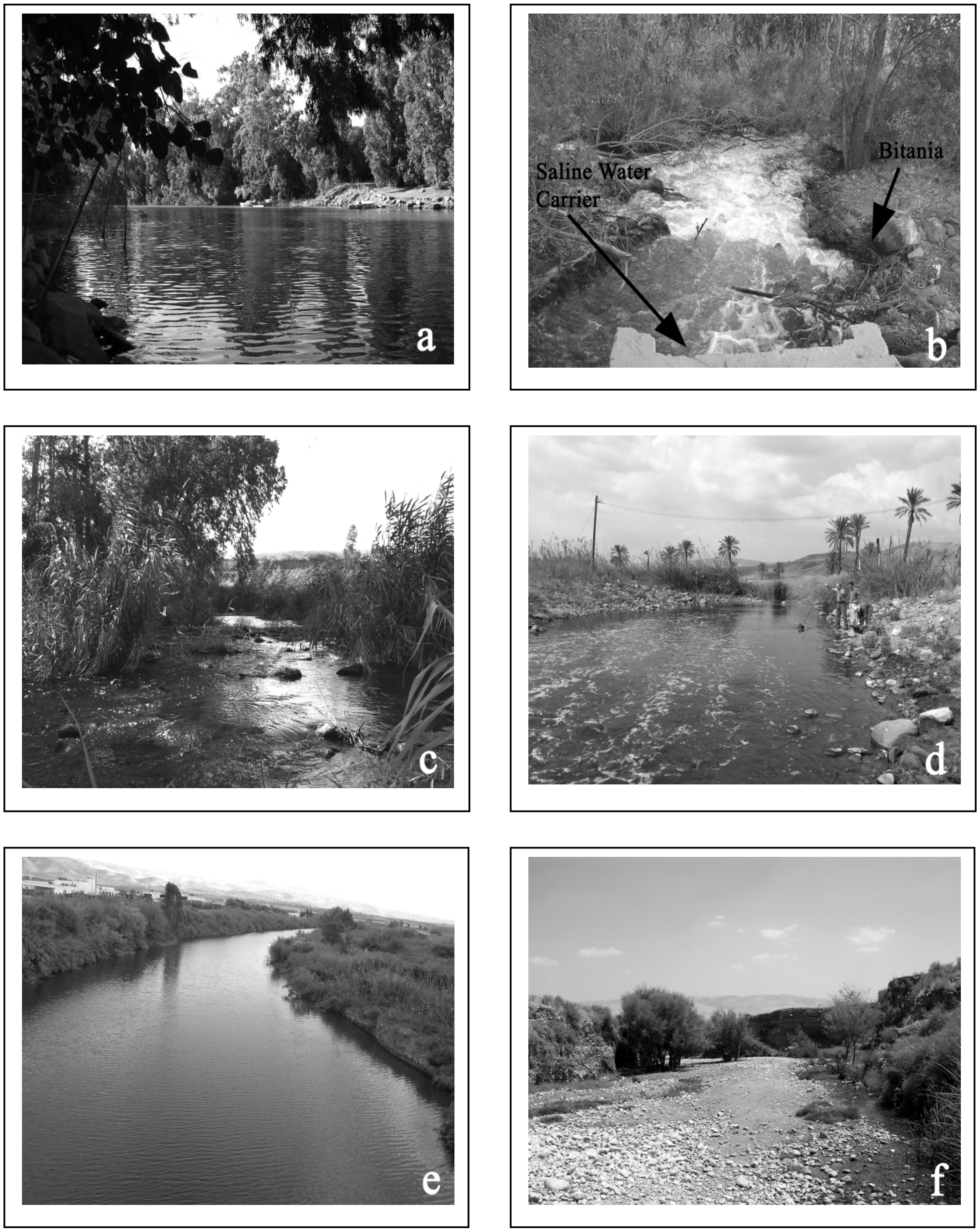

Figure 2. *Sampling stations over the Lower Jordan River and tributaries: a – st. 3; b – st. Bitania and Salinity Water Carrier; c – st. 5; d – st. 7; e – st. 8a; f – tributary Bezet*

Materials and methods

For our study we collected 34 samples of planktonic and periphytonic algae from the Lower Jordan River at the end of dry season in November 2005 and at the end of wet season in May 2007. The samples were collected at 13 accessible sampling stations along the Lower Jordan River and its tributaries (*Figures 1, 2*). The samples were obtained by scooping up for phytoplankton and by scratching for periphyton and fixing in 3% formaldehyde. The algae were studied with a dissecting Swift microscope under magnifications of 740-1850. The diatoms were prepared with the peroxide technique (Swift, 1967) modified for glass slides (Barinova, 1997). In parallel with the sampling for algae we collected water samples for chemical analyses that are regularly performed by the Ministry of Environment of Israel (*Table 1, 2).*

Table 1. *Environmental variables along the Lower Jordan River in November 2005, Ec – Electrical conductivity (mSm/cm), E. coli – cells per 100ml; all other variables in units of mg/l.*

2005	**km**	**pH**	**T°C**	**Ec**	**BOD**	**COD**	**Cl**	**N-kel**	**T.P.**	**N-NO_3**	**N-NO_2**	**NH_4**	**T.N.**	***E. coli***
1	0	8.23	21.6	1.08	2.9	44	287	0.9	<0.2	<1	0.001	0.2	0.9	3100
3	1.7	7.53	18	1.41	1.9	52	317	1.3	<0.2	<1	0.02	0.3	1.3	1300
3a	2.0	7.42	24.3	6.52	17	76	2118	9.9	3	<1	0.16	7.1	10.1	280000
5	4.1	7.49	22.4	5.95	24	88	1858	7.8	2	<1	1.1	5.8	8.9	150000
7	10.9	7.54	19.8	6.41	21	75	2054	7.7	3.1	<1	1.09	6	8.8	75000
8	15.9	7.64	19.6	5.82	10	36	1858	2.8	2.3	3.4	1.6	1.3	7.8	2100
8a	40.3	7.87	19.3	6	15	72	1892	7.8	2.2	<1	0.71	1.9	8.5	7300

Table 2. *Environmental variables along the Lower Jordan River in May 2007, Ec – Electrical conductivity (mSm/cm), E. coli – cells per 100ml; all other variables in units of mg/l.*

2007	**km**	**pH**	**T°C**	**Ec**	**BOD**	**COD**	**Cl**	**N-kel**	**T.P.**	**N-NO_3**	**N-NO_2**	**NH_4**	**T.N.**	***E. coli***
1	0	6.60	27	0.82	0.4	81	241	0.9	<0.2	<0.2	0.01	<0.1	0.9	80
3	1.7	6.90	26	0.96	12	50	269	2.9	0.4	<0.2	0.018	<0.1	2.9	170
3a	2.0	6.75	28	6.47	20.7	72	1885	17.5	4.4	<0.2	<0.01	12	17.5	110000
5	4.1	7.20	26	4.63	16.5	62	1949	15.9	3.6	<0.2	0.1	9.3	16	150000
7	10.9	7.45	28	5.95	30	132	1914	13.7	4.2	<0.2	0.93	4.2	14.6	11000
8	15.9	7.50	-	5.93	18	86	1914	7.2	6.1	4.4	1.42	1.3	13	1400
8a	40.3	-	-	-	15	102	1659	5.7	1.9	3.4	0.96	1.4	10	1700

The taxonomy of this study mainly follows the systems adopted in the "Süswasserflora von Mitteleuropa" (Ettl, 1978; Starmach, 1983, 1985; Ettl and Gartner, 1988; Krammer and Lange-Bertalot, 1991a,b,c,d; Komárek and Anagnostidis, 1998), and Green Algae on K.R. (Mattox and Stewart, 1984), with additions for individual taxa (Gollerbach et al., 1953; Desikachary, 1959; Krammer, 1985, 2000; Moshkova and Gollerbach 1986; Lange-Bertalot and Krammer, 1987; Meffert, 1987; Komárek and Anagnostidis, 1989; Popovsky and Pfiester, 1990; Perestenko, 1994; Barber and Carter, 1996; Lenzenweger, 1996; Hegewald, 2000; Rumrich et al., 2000).

Ecological characteristics of the species are compiled in our database (Barinova et al., 2006a). Our ecological analysis revealed the groups indicator species for pH, salinity, and saprobity level. Each group was separately assessed according to its significance in bio-indication of eutrophication, pH levels (acidification), salinity, and organic pollutants. The density scores were calculated using the 6-score scale (Korde, 1956) for the saprobity index S, as well as the 5-score scale (Whitton at al., 1991) for the saprobity index EPI.

Saprobity Index (S)

The Saprobity Index (S) was calculated as:

$$S = \sum_{i=1}^{n}(s_i . a_i) \Big/ \sum_{i=1}^{n}(a_i) \qquad \text{(Eq. 1)}$$

where S – the index of saprobity for the algal community;
s_i – the species-specific saprobity index;
a_i – the density score.

S value between 1 and 4 is the "weighted mean" of all individual indices that defines the self-purification zone corresponding to five classes of water quality (Sládeček, 1973).

This bio-indication approach is based on the ecological classification, which is widely used in European and Asian countries (Romanenko et al., 1990; Whitton et al., 1991; WFD, 2000).

Environmental Pollution Index (EPI)

An alternative approach to biological assessment of water quality is provided by the diatom-based index of eutrophication and pollution, EPI (Dell'Uomo, 1996) which shows a significant correlation with the chemical and physical properties of the water such as the levels of BOD5, nutrients, conductivity, chlorides, phosphates, etc. In this approach, the indicator species were selected on account of their nutrient level tolerance (Trophic Degree: Dell'Uomo, 1996) and five different organic pollution levels (Sládeček, 1973).

The Environmental Pollution Index (EPI) is calculated as:

$$EPI = \sum_{j=1}^{n} a_j r_j i_j \Big/ \sum_{j=1}^{n} a_j r_j \qquad \text{(Eq. 2)}$$

where EPI – eutrophication/organic pollution index for the station;
a_j – species abundance by the 5-score scale;
r_j – constant species index of eutrophication/pollution, EPi;
i_j – individual coefficient R for the diatom indicator list (Dell'Uomo, 1996), ranging form 1 to 5.

The index EPI ranges between 0 and 4, and negatively correlated to such aquatic ecosystem variables as halobity and the trophic level (Dell'Uomo, 1995). The water quality Classes of a sampling station can be estimated on the basis of the inter-correlated variables summed up in *Table 3*.

Table 3. *Correlation of the saprobity, halobity, trophic level, index EPI and the water quality classes (Dell'Uomo, 1996).*

Saprobic degree	Trophic degree	Halobic degree	EPI range	Class of water quality	Water quality
xenosaprobic	hypotrophic	halophobous	0.0 – 0.5	0	Natural, unpolluted water
oligosaprobic	oligotrophic	oligohalobous-indifferent	0.5 – 1.0	I	Excellent water quality
			1.0 – 1.5		Good water quality
β-mesosaprobic	mesotrophic	oligohalobous - indifferent	1.5 – 2.0	II	Fairly good water quality
			2.0 – 2.5		Slightly polluted water
β-mesosaprobic	eutrophic	oligohalobous - halophilous	2.5 – 3.0	III	Rather polluted water
			3.0 – 3.5		Strongly polluted water
polysaprobic	hypertrophic	halophilous - mesohalobous	3.5 – 4.0	IV	Heavily polluted water

The second alternative method of water quality assessment is based on three groups of indicators: saproxenes of clean water, eurysaprobes of medium quality water, and polysaprobes of polluted water defined by Watanabe et al. (1986).

The Integral Index of River Pollution (RPI)

The integral index of river pollution (RPI; Sumita 1986) was calculated for the entire river flow, based on the DAIpo (Diatom Assemblage Index of Organic Pollution) for each of the sampling stations. This index has been first applied for the rivers of Japan (Watanabe et al. 1986). We adopted RPI for estimating a number of environmental variables of the river flow, based on conductivity, pH, nitrate concentrations, etc. The integral indices (Tavassi at al. 2004; Barinova et al. 2006a) were calculated as:

$$RPId = \sum_{i=1}^{n}(Di+Dj)*l \Big/ 0.5L \qquad \text{(Eq. 3)}$$

where Di, Dj - an estimate of environmental variable or the corresponding index value for adjacent stations i and j;

l - the distance between two adjacent stations (km);

L - the total length of the river.

The Aquatic Ecosystem State Index (WESI)

The index of ecosystem status (Aquatic Ecosystem State Index, WESI) is based on the water quality classes (Barinova, et al. 2006a) reflecting the self-purification capacities for each of the sampling stations. It is calculated as:

$$\text{WESI} = \text{Rank S / Rank N-NO}_3 \qquad \text{(Eq. 4)}$$

where: Rank S – rank of water quality based on the Sládeček indices of saprobity.

Rank $N\text{-}NO_3$ – rank of water quality based on the nitric-nitrogen concentration.

If WESI is equal to or larger than 1, the photosynthetic level is positively correlated with the level of nutrients concentration. WESI is less than 1 would signify that photosynthesis was suppressed, presumably in the consequence of a toxic impact.

Not only the nitric-nitrogen concentrations, but also the other ecologically significant environmental variables that impacted algal community can be used as the denominator of the WESI equation 4.

Bioassay of Bacteria species

The ToxScreen Toxicity Test was purchased as a kit from the CheckLight Company (CheckLight Ltd. P.O.Box 72 Qiryat-Tivo'n 36000, Israel). It included lyophilized bacteria, *Photobacterium leiognathi*, Hydration buffer, Pro-Organic buffer (which favors the detection of organic pollutants), and Pro-Metal buffer (which favors the detection of heavy metal cations). These buffers were developed to enhance the sensitivity of the test to a wide range of agents with different modes of action. The chosen concentrated Assay Buffer (Pro-Metal or Pro-Organic) was diluted in ultra-pure water in which serial dilutions of the tested toxic agent were performed in polystyrene vials. Next, a vial containing the lyophilized luminous bacteria was removed from the freezer, re-hydrated with 1ml of Hydration Buffer and incubated at 24 °C for 5 minutes. A suspension of 10µl of cells was dispensed into each vial and thoroughly mixed. Vials were incubated at 24 °C for 30 minutes, and luminescence was recorded. The bioluminescence index (BI) was determined by the ratio of the bacterial luminescence of water sample from the Lower Jordan River compared to the clean water control sample.

Statistical data analyses

Statistical analysis was conducted using Statistica 7.0 (StatSoft, 1996). The relationship of diversity data (species richness of a community and of each major taxonomic division Cyanoprokaryota, Bacillariophyta and Chlorophyta that was represented in each community, as well as saprobity index S) with environmental data on the sampling stations estimated by stepwise multiple regression analysis, where numbers of species were used as dependent and the environmental data as independent variables.

Result and discussion

Benthic algal communities are usually species-rich and therefore represent an information-rich assemblage ideal for environmental monitoring because of their relatively short life cycles, allowing a rapid response to environmental impacts, at the same time revealing a cumulative effect of consecutive environmental events a longer period of time (Janauer and Dokulil, 2006). Because of this, algal bio-indication is included in the EC-Water Framework Directive (WFD, 2000) for water quality assessment and monitoring.

Water quality conditions

There is an abrupt increase of electric conductivity, chloride, BOD, ammonium, *E. coli,* and in total phosphorus and nitrogen in the lower section (after the Alumot-dam) relative to the upper section (*station 1 and 3),* which is blocked by the Alumot-dam (*Tables 1, 2*). This increase could be related to the effluents of Saline Water Carrier and the Bitania wastewater treatment plant, which are the major water sources at the inlet of the Lower Jordan River after the Alumot-dam. As a whole, the Lower Jordan River is alkaline, temperate, fresh in the upper section, and brackish in the lower section. According to the water quality estimates, the upper section can be considered unpolluted while the lower section has a high level of BOD, nitrogen, phosphorus, etc., and is considered heavily polluted.

Taxonomy

We recognized 105 species of algae from nine taxonomical divisions in November 2005; and 86 species of algae from six taxonomical divisions in May 2007. Altogether, for both years we identified 152 species (Tavassi et al., in press) in 34 samples of plankton and periphyton collected from 13 stations over the Lower Jordan River and its tributaries (*Table 4*). During the study period, the diatoms (Bacillariophyceae) were accompanied by the greens (Chlorophyta) and blue-greens (Cyanoprokaryota) with the euglenoids (Euglenophyta) next in abundance. In 2005, the chrysophytes (Chrysophyta), Xanthophyta, Dinophyta, Cryptophyta, and Rhodophyta were represented by only one or two species each, while in 2007 Dinophyta were represented by 3 species and Cryptophyta by one, which reflected the differences in species composition in dry and wet seasons: in the wet season the diversity is higher. Of 152 species, 135 (89%) are indicators of environmental conditions such as habitats, temperature, streaming and oxygenation, saprobity, halobity, and acidification.

Table 4. *Taxonomical distribution of algal indicators diversity over the stations of the Lower Jordan River*

Station at the cannel	**1**	**3**	**3a**	**5**	**7**	**8**	**8a**
Distance, km	0	1.7	2	4.1	10.9	15.9	40.3
Cyanoprokaryota	8	10	4	2	2	9	3
Bacillariophyceae	4	10	3	3	9	8	3
Chlorophyta	14	15	2	2	1	5	9
Chrysophyta	1	1	0	0	0	0	0
Dinophyta	1	1	0	0	0	1	1
Euglenophyta	1	2	0	0	1	0	0
Cryptophyta	0	1	0	0	0	0	0
Rhodophyta	0	0	0	0	0	1	0
No of Species	29	40	9	7	13	24	16

The dynamic of algal divisions over the stations (*Table 4*) shows a significant decrease in both high taxa and species diversities in the lower section, from seven divisions with 40 species at station 3 (above the Alumot-dam) to only three divisions with 9 species at station 3a (below the Alumot-dam). These dynamics represent the marked differences between the upper and lower sections, which persisted round the year (*Figure 3*).

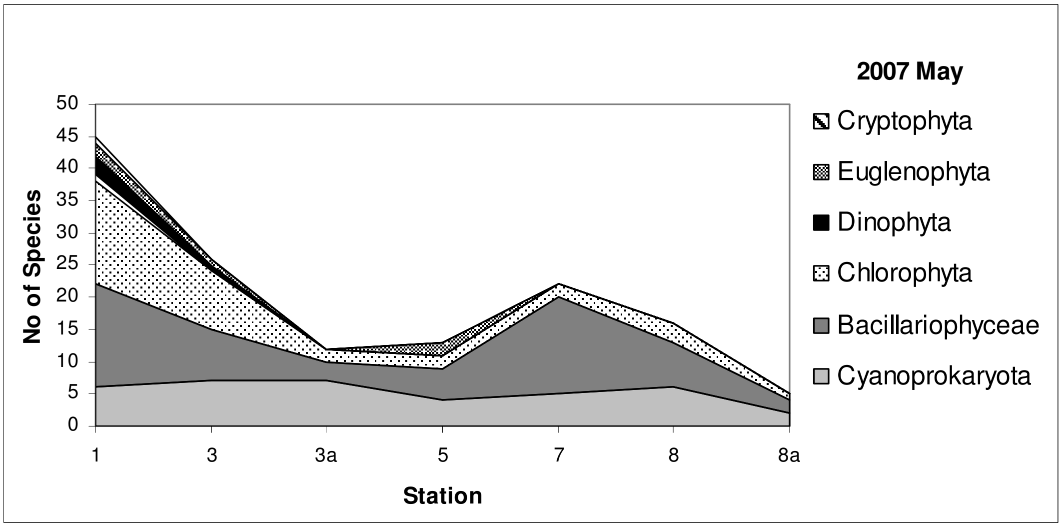

Figure 3. *Distribution of taxonomical diversity over the sampling stations of the Lower Jordan River*

Chloride concentrations (reflecting salinity load) measured at all sampling stations (*Table 1, 2, Figure 4*), show that the river is fresh water in the upper section (*stations 1 and 3, Figure 2a)* and highly brackish in the lower section, after the Alumot-dam, where the Saline Water Carrier and Bitania wastewater treatment plant are discharging water to the river. *Figure 5* shows distribution of algal bio-indicators along the Lower Jordan River in relation to chloride concentrations. The upper section, at *stations 1* and *3*, is dominated by freshwater species, 97% and 92%, respectively. However, in the lower section (after the Alumot-dam), at *stations 3a* (Figure 2b) and *5 (Figure 2c*), there is a sharp decrease in freshwater species (to 57%), accompanied by a decrease in the number of species (*Figure 3*). A recovery of the total algal diversity and the numbers of freshwater species was observed at station 7, 10.9 km after the Alumot-dam.

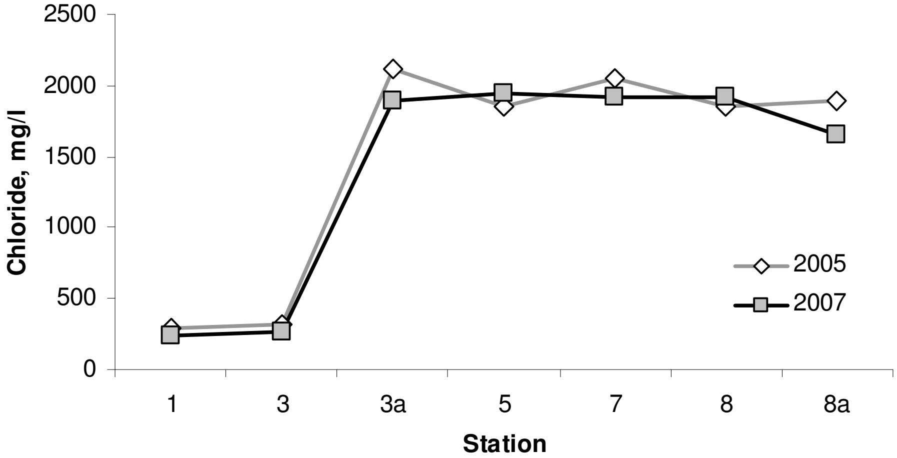

Figure 4. *Seasonal Chloride fluctuations over the Lower Jordan River stations*

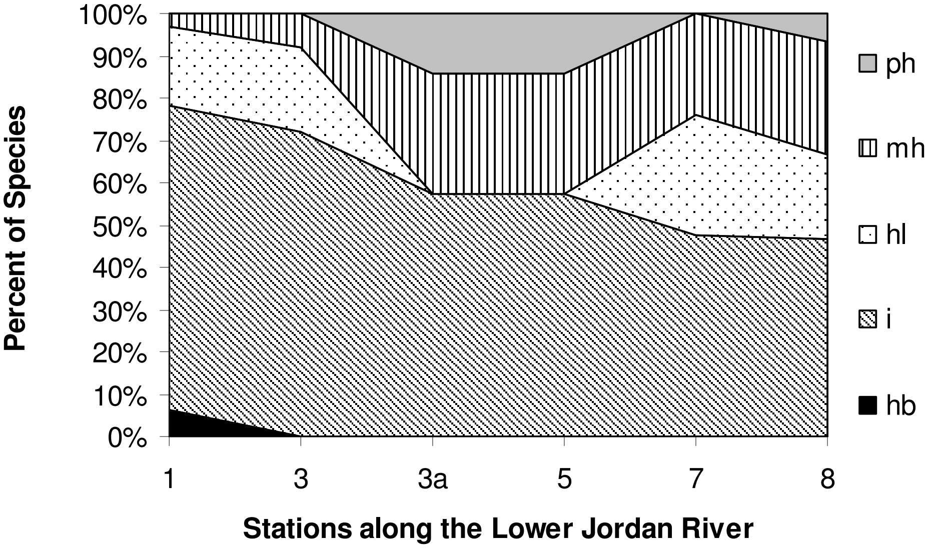

Figure 5. Dynamic of algal species indicators of salinity over the Lower Jordan River stations

The indicators of salinity (86 species, 56.6%) are assigned to five ecological groups based on the gradient of chloride concentration. The majority of them are "indifferents" (45 species) of a broad tolerance of salinity fluctuations. These include such abundant species as *Nitzschia palea, Navicula recens, Plagiotropis lepidoptera* (Bacillariophyta), and *Cladophora glomerata* (Chlorophyta). The less prominent halophiles and mesohalobes were represented by 17 and 18 species, respectively. Remarkably, the algal communities of the Lower Jordan River also contain polyhalobes that prefer marine or brackish waters of a high-chloride concentration. They include *Microcoleus chthonoplastes* (Cyanoprokaryota) and *Enteromorpha torta* (Chlorophyta), which were found at the lower section (after the Alumot-dam). The group ratios suggest a wide range of salinity.

Saprobity

The indicators of organic pollution are assigned to 13 groups of Pantle-Buck's (1995) system. They included 60 species (57%) in 2005 and 57 species (66%) in 2007. In 2005, the most representative among them were the main groups of oligo-, oligo-beta- and beta-mesosaprobionts (7, 8, and 16 species, respectively), while in 2007 the main groups were oligo- and beta-mesosaprobionts (9 and 13 species, respectively). The transitional groups were less prominent. Such a distribution attests to a high heterogeneity of the Lower Jordan River Basin in respect of organic pollution. Ecological dominants for different groups of saprobionts are: *Navicula viridula*, *Bacillaria paxillifer* and *Caloneis amphisbaena* (Bacillariophyta) among the oligosaprobionts, *Navicula recens* (Bacillariophyta) and *Lyngbya kuetzingii* (Cyanoprokaryota) among the oligo-beta-saprobionts, and *Phormidium ambiguum*, *Aphanizomenon flos-aquae* and *Phormidium autumnale* (Cyanoprokaryota) among the beta-saprobionts. *Figure 6* shows the dynamics of algal indicators of organic pollution over the Lower Jordan River. In the upper section, at *stations 1 and 3*, the dominant species are indicators of a good to moderate water quality Classes. The indicators of pure water are represented by only one, two, or three species at each station. However, at stations 3a and 5, we observed a sharp decrease in number of good to moderate quality indicators. Their recovery coincided with that of the total diversity and salinity indicators at station 7, 10.9 km after the Alumot-dam (*Figure 2d*).

Figure 6. *Dynamics of algal indicators of organic pollution over the Lower Jordan River stations*

Indices S

The saprobity indices calculated for each of the sampling stations (*Equation 1*) reflect organic pollution levels that vary from 1.46 to 2.14 in the rainy season and from 1.39 to 2.19 in the dry season (*Figure 7*). The amplitude of variation falls in the rank of the oligo- to betamesosaprobic self-purification zones attesting to the water quality Class III of moderately polluted waters at the outlet decreasing to Class II at the lowermost river *stations 7* (*Figure 2d*) *and 8.* The range of variations decreases down the river, peaking at the station 3a, which indicates a contribution of wastewaters from Bitania and the Salinity Water Carrier. The pollution levels are relative stable in the upper section, while the higher fluctuations were found in the lower section, almost 40 km long, with an appreciable to self-purification tendency. In the tributaries of the Lower Jordan, the indices of saprobity S widely varied: 1.79 in Bitania, 1.33-1.58 in Salinity Water Carrier outflow, 1.51-1.80 in Yarmuk, 0.8-1.53 in Tavor, 1.71-2.49 in Harod, 1.34-1.47 in Bezet. In the tributaries the haviest pollution was observed during the rainy season. The Harod River that flow to the Lower Jordan River from the Yehudea Mountain was dramatically polluted by the outwash from the sheep's pasture in May, 2007.

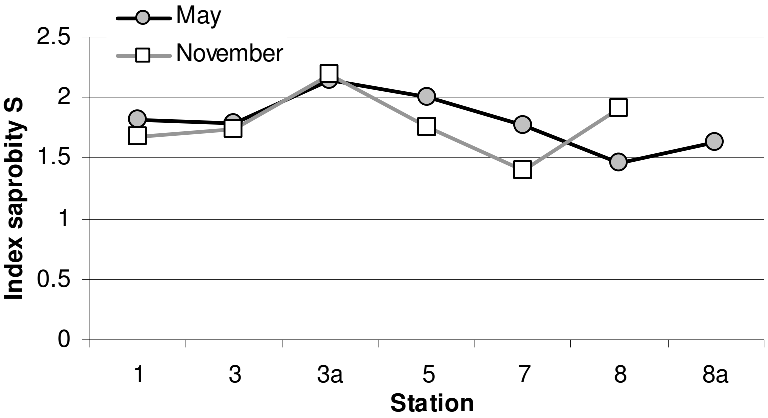

Figure 7. *Seasonal dynamics of saprobity indices S over the Lower Jordan River stations*

Indices EPI

The same bio-indication methods were used for the Kishon, Yarqon, Hadera and Alexander rivers in northern Israel (Barinova et al., 2004, 2005, 2006b,c; Tavassi et al., 2004). For comparison with the other Mediterranean regions we used Dell'Uomo's (1995) calculation of EPI (*Equation 2*). The results are presented in *Figure 8* showing the EPI's variation from 1.0 to 4.5 that indicate unpolluted to heavily polluted waters. The algal communities reveal the oligosaprobic to polysaprobic self-purification ranks and the oligotrophic to hypertrophic levels (Dell'Uomo, 1995; *Table 3*). The trend lines of both seasonal distributions are similar (*Figure 8*) and reveal two different parts of the canal – the upper, above the Alumot-dam, and the lower below it. As a whole, the fluctuations of EPI's are unconformable to those of S, the ranges are wider and the maxima are displaced in the direction of the heavier polluted alfameso- to polysaprobic waters. Even greater discrepancies were found between S and EPI in the case of pollution estimates. The EPI does not discriminate between the pollution levels at the upper *station 1* and the lower *stations 5, 7 and 8*, which differ from all the other estimates, at the some time not responding to a sharp increase in pollution immediately after Bitania wastewater inflow.

On the other hand, the number of indicator species that are included in the equation of EPI was too low – from 0 to 4 for each community, which makes the results unconvincing in our case. The EPI is based on the diatoms alone, whereas the index S involves a greater diversity of algal groups and is, therefore, more reliable. We come to similar conclusions in the case of the Yarqon River communities (Tavassi et al., 2004).

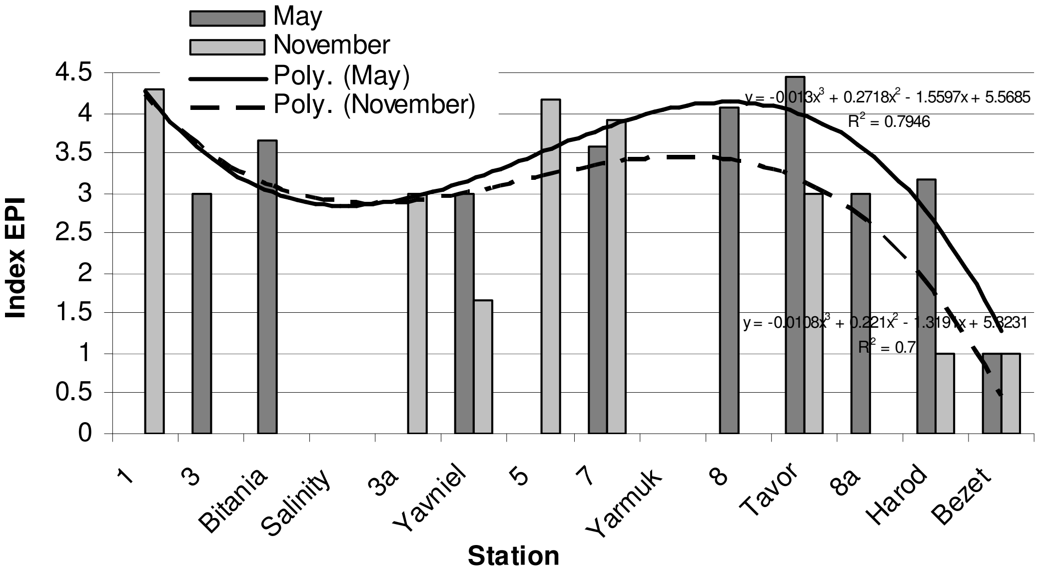

Figure 8. *Seasonal dynamics of saprobity indices EPI over the Lower Jordan River and tributaries*

RPI indices

Table 5 presents the results of our calculations of RPI (*Equation 3*), after Sumita (1986) and Barinova et al. (2006a), for the Lower Jordan River. The RPI includes organic nitrogen (N-kel.), BOD, COD, chloride (Cl), total phosphorus (P), nitrate (NO_3), nitrite (NO_2), ammonium (NH_4), and total nitrogen (N). The RPI values in relation to N-kel., COD, P, and N were high in May 2007 compared to those found in November 2005. These data may indicate a decrease of water quality in the wet season caused by influx of pollution from the basin area. However, in relation to Cl, the RPI values were more stable over the seasons. The major source of chloride is the Saline Water Carrier. Based on RPI stability we conclude that the volume of salt water is nearly the same for the dry and rainy seasons.

Table 5. *River Pollution Indices (RPI) based on chemical analysis of the Lower Jordan River*

RPI	N-kel.	BOD	COD	Cl	T.P.	N-NO_3	N-NO_2	NH_4	T.N.
November 2005	6.36	17.64	69.48	1770	2.30	1.38	0.98	4.62	7.87
May 2007	12.22	20.92	92.76	1729	3.89	0.86	0.60	5.28	13.49

Statistical analysis

For the Multiple Regression Analysis we clustered the environmental variables into three major groups. Group 1 includes the macro-variables pH, T°C (T), Ec, Cl, NH_4. Group 2 includes the variables, which show the of ecosystem processes BOD, COD,

NH_4, T.N. (TN) E. coli. Group 3 includes trophic variables N-kel.(Nk), T.P. (TP), N-NO_3 (NO_3), N-NO_2 (NO_2). The Multiple Regression Analysis indicated a greater range of fluctuations in the algal community relative to the environmental variables (*Table 6*).

Table 6. *Multiple regression coefficients (R^2) of algal community parameters as dependent variables with environmental variables as independent factors in the Lower Jordan River stations in 2005-2007 (p: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001)*

	Group 1: pH, T°C (T), Ec, Cl, NH_4				
	Step 1	Step 2	Step 3	Step 4	Step 5
Cyanoprokaryota	-	-	-	-	-
Bacillariophyceae	-	-	-	-	-
Chlorophyta	Cl 0.77***	Cl T 0.83***	Cl T NH_4 0.82***	pH T Cl NH_4 0.81***	-
No of Species	Cl 0.69***	Cl T 0.70***	Cl T pH 0.69**	Cl T pH NH_4 0.67***	-
Index S	-	-	-	-	-
	Group 2: BOD, COD, NH_4, T.N. (TN), *E. coli*				
	Step 1	Step 2	Step 3	Step 4	Step 5
Cyanoprokaryota	COD 0.25*	-	-	-	-
Bacillariophyceae	-	-	-	-	-
Chlorophyta	TN 0.74***	T.N. BOD 0.80***	TN BOD *E.coli* 0.79***	TN BOD NH_4 *E.coli* 0.81***	TN BOD NH_4 *E.coli* COD 0.81**
No of Species	TN 0.59***	-	-	-	-
Index S	*E.coli* 0.29*	-	-	-	-
	Group 3: N-kel. (Nk), T.P. (TP), N-NO_3 (NO_3), N-NO_2 (NO_2)				
	Step 1	Step 2	Step 3	Step 4	Step 5
Cyanoprokaryota	-	-	-	-	-
Bacillariophyceae	-	-	Nk TP NO_3 0.43*	-	-
Chlorophyta		Nk NO_2 0.76**	Nk NO_2 NO_3 0.77***	Nk NO_3 NO2 TP 0.78**	-
No of Species	Nk 0.41**	Nk NO_3 0.60**	Nk NO_3 TP 0.64**	Nk TP NO_3 NO_2 0.66**	-
Index S	-	-	-	-	-

The most significant correlation is established for the species richness of entire communities, in particular the diversity of green algae, with salinity (negative) and the total and organic nitrogen concentrations (positive). The effect of temperature, nitrates, ammonia and pH is less significant, but still recognizable. The blue-green algae are influenced by COD, whereas the impact of other variables is insignificant. The diatom diversity depends on the trophic conditions (positive) in the first place. The saprobity index S is correlated with the abundance of *E. coli* alone (positive). But it is well known that *E. coli* survives in the organically polluted water.

The Aquatic Ecosystem State Index (WESI)

WESI conveys a self-purification capacity of aquatic ecosystems. *Figure 9* represents the results of WESI calculation (*Equation 4*) for the Lower Jordan River. The variables we used in this case are the saprobity index S and the ammonia concentration rate for each canal station. During both seasons, the WESI shows similar dynamics. At the upper *stations 1 and 3*, the WESI remained above one round the year, which led us to conclude that the river ecosystem was sustainable within the normal ranges of self-purification. In contrast, in the polluted section of the river canal between *stations 3a and 8a*, the WESI was constantly below one indicating a suppressed self-purification process.

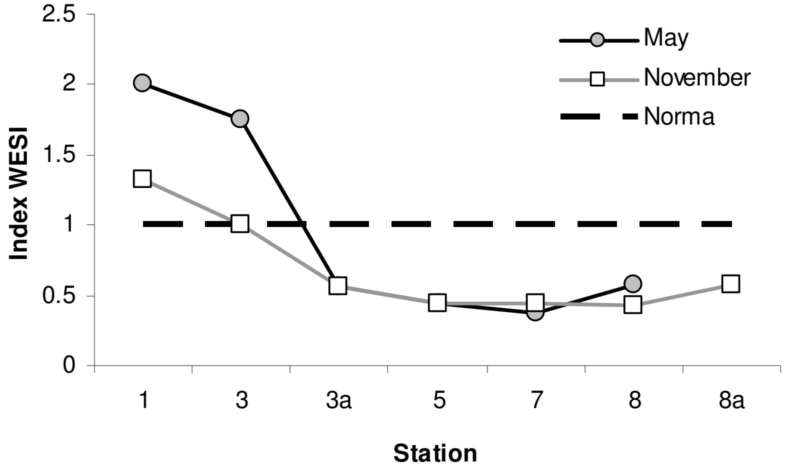

Figure 9. *Seasonal dynamics of integral ecosystem indices WESI over the Lower Jordan River stations*

Bioluminescence indices (BI) of toxicity

A highly sensitive stain of a luminescent bacterium, *Photobacterium leiognathi,* was used to measure water toxicity level of the Lower Jordan River. The action of water pollution on the bacterial luminescence was estimated with the bioluminescence index (BI). In the Pro-Metal buffer (favoring detection of heavy metal cations), the BI value increased up to 200% compared to the control sample. Evidently the Pro-Metal buffer renders the bacteria too sensitive to organic matter in the water and therefore of little utility in estimating the toxicity levels. Better results we obtained with the Pro-Organic buffer (favoring detection of organic pollutants), for which a treatment of luminous bacteria with water from the upper section (*stations 1 and 3*) had a slight influence on bioluminescence intensity, with the BI values 0.67-0.81. Below Bitania and the Saline Water Carrier (*stations 3a – 8*) the BI values 0.38-0.53 indicated an increase toxicity level.

Figure 10 demonstrates correlation between BI and Electrical conductivity. Our results suggest that inhibition of bacterial luminescence may reflect the level of pollution by the waste and saline waters from Bitania and the Saline Water Carrier. Moreover, our results imply that the level of inhibition can be used for classification of water quality. The BI > 0.5 indicates a good water quality, whereas the range of 0.5 < BI < 0.1 corresponds to a fair water quality.

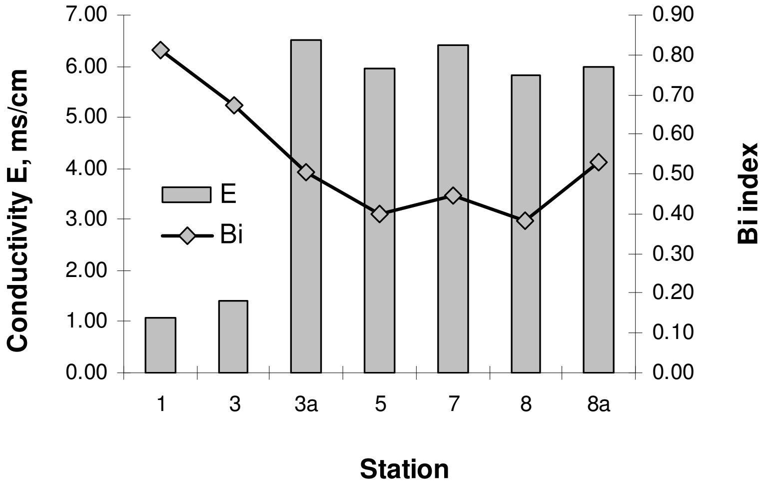

Figure 10. *Comparison of toxicity indices BI and Electrical conductivity over the Lower Jordan River stations*

Nearly identical BI values were obtained for *stations 1 and 3* of the main canal and for the tributaries Yavniel, Tavor, and Yarmuk, whereas a fair water quality was estimated for main canal from Bitania and the Saline Water Carrier till *station 8*. Self-purification resumes only at the *station 8a* down the river.

A comparison of BI and saprobity indices S over the river canal stations shows (*Figure 11*) that that indices are a negative correlated: an increase in S below Bitania is accompanied by an appreciable decrease of BI. And vice versa, a self-purification trend of the lower reaches corresponds to a considerable decrease of S (trend lines in *Figure 10*).

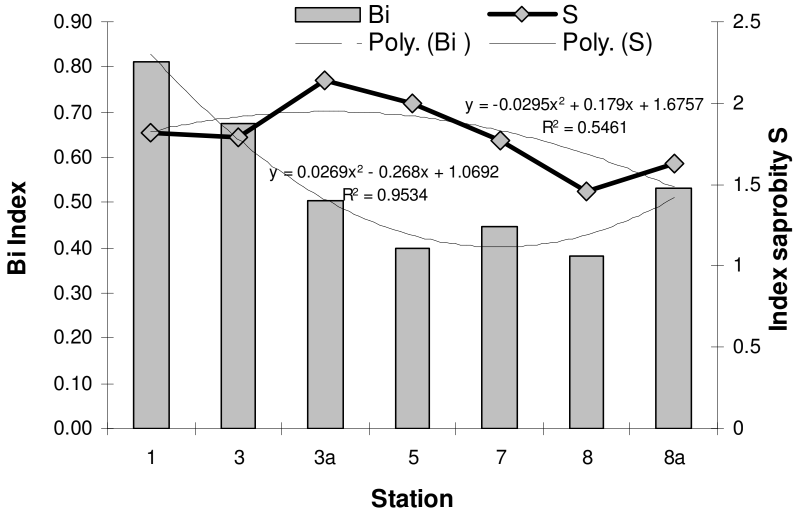

Figure 11. *Comparison of saprobity index S and toxicity index BI over the Lower Jordan River stations*

Conclusion

On the basis of bio-indication analysis, the Lower Jordan River is characterized as alkaline, temperate, with fresh water in the upper reaches (below the Alumot-dam), and brackish in the lower reaches. Pollution is insignificant in the upper reaches, but the high levels of BOD, nitrogen, and phosphorus indicate a dramatic increase in pollution down the river.

During the study period (November 2005 & May 2007) we identified 152 algal species of plankton and periphyton from seven taxonomic divisions. The algal flora is dominated by the diatoms (*Bacillariophyceae*), with the greens (*Chlorophyta*) and blue-greens (*Cyanoprokaryota*) next in abundance, and followed by the euglenoids (*Euglenophyta*). The other divisions are represented by only one or two species each. The differences in the species diversity observed in 2005 and 2007 reflect the seasonal changes. Diversity is higher in the dry season. We selected 135 species (89%) as indicators of environmental conditions. Bio-indicators of chloride concentrations divide the river flow into two sections: one above Alumot-dam in inhabited by freshwater species, whereas polyhalobes appear below the dam indicating an influx from the Saline Water Carrier and Bitania. The typical polyhalobes are *Microcoleus chthonoplastes* (*Cyanoprokaryota*) and *Enteromorpha torta* (*Chlorophyta*).

The saprobity index S, which reflects organic pollution, varied from 1.46 to 2.14 in the dry season and from 1.39 to 2.19 in the rainy season. In the upper section, the pollution levels are low and relatively stable, while the grater fluctuations with a tendency of self-purification, were observed over the 40 km length of the lower section. The Mediterranean saprobity index EPI was of little utility in this case because the relatively low numbers of indicator species in the Lower Jordan River communities

The RPI values, determined for both seasons, indicate a higher pollution in the wet season compared to the dry season, the differences being related to washout by rain waters. However, the RPI value for chlorides was relatively stable in both seasons.

The indices of ecological status WESI markedly differ for the two parts of the river canal. At the upper unpolluted stations, the self-purification processes are quite active, but decrease below the Alumot-dam where the algal communities are impacted by pollution.

The bioluminescence index (BI) is strongly correlated with the Electrical conductivity and chlorides reflecting the combined action of salinity and organic pollutants, the most important environmental variables for the algal assemblages of the Lower Jordan River. The BI clearly shows the crucial role of the Salinity Water Carrier and Bitania as the major sources of pollution in the river.

The Statistical Multiple Regression Analysis confirmed the significance of salinity as a major impacted factor for the total species richness of the algal communities. The diatoms and greens are mainly positive influenced by trophic conditions, but the green algae are no less sensitive to temperature and pH. The development of cyanobacteria is mainly controlled by COD. The saprobity index S is correlated with the abundance of *E. coli*, which survives in the organically polluted water.

We consider the evaluation of bio-indication indices of water quality as a preparatory stage for a complex biological monitoring of the Jordan River, one of the major trans-boundary water sources of the East-Mediterranean region. Biological monitoring of human-environment interactions (Dearing et al., 2006) involves a number of bio-indication methods and statistic analysis of both natural and anthropogenic variables impacting the self-purification capacity of aquatic communities.

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FUZZY AND BOOLEAN FOREST MEMBERSHIP: ON THE ACTUAL SEPARABILITY OF LAND COVER CLASSES

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Abstract. Forests are among the most important habitats of the Earth for several ecological reasons and their management is a prior task when dealing with landscape conservation. Thematic maps and remote sensing data are powerful tools to be used in landscape planning and forest management; nevertheless, most of the European and Mediterranean forest monitoring and conservation programs do not take into account the continuity of the variation of habitats within the landscape but they only rely on boolean classification methods. The utilisation of a classification method that applies a continuity criterion is fundamental because it is expected to better represent the ecological gradients within a landscape. The aim of this paper is to assess the amount of classification uncertainty related to crisp (boolean) classes, particularly focusing on forest identification uncertainty. Forest fuzzy membership of the Tuscany region (Italy) derived from a Landsat ETM+ image scene was compared with the widely used crisp datasets in European forests management and conservation practices, i.e. the European JRC Forest/Non-Forest map, the CORINE Land Cover 2000 (levels 1 and 2), as well as the Global Land Cover 2000, in order to qualitatively and quantitatively assess the separability of crisp classes with respect to forest fuzzy membership. A statistically significant ($p < 0.001$) forest fuzzy membership separability among the considered crisp classes was found. Despite the crisp dataset and hierarchical level taken into account, both forest and non-forest crisp classes showed a high degree of forest fuzzy membership variability. Therefore, given the intrinsic mixture of crisp land cover classes, ecological studies on forestal ecosystems should rigorously take into account the classification uncertainty related to a crisp view of ecological entities which are being mapped.

Keywords: *biodiversity, classification uncertainty, forest conservation, forest management, fuzzy set theory, remote sensing*

Introduction

Forests represent one of the most important habitats on Earth for several ecological reasons, including: hosting a great amount of Earth's biodiversity (Sohngen et al., 1999); preventing soil erosion; replenishing ground water by reducing water runoff; controlling flooding; enhancing infiltration; and storing carbon (Perry 1994; Oren et al., 2001).

Forest fragmentation represents one of the crucial phenomena responsible for global decline of biodiversity (Wilcox and Murphy 1985), particularly in the Mediterranean biome where a high amount of native forest has been converted (Hoekstra et al., 2005). In order to protect forests, several international programs were drawn up in The Convention on Climate Change, Convention on Biological Diversity and Ministerial Conference on the Sustainable Forest Management (McRoberts and Tomppo, 2007).

As firstly recognised by Wulder (1998) and further reinforced by McRoberts and Tomppo (2007), the efficiency of forest management could be improved if forest planners and managers would apply remote sensing data. In this view, thematic maps (e.g. land cover, forest cover, forest habitats) are being increasingly used in landscape planning and management (Kepner et al., 2000; Butler et al., 2004; Romero-Calcerrada and Perry, 2004; Acosta et al., 2005; Rocchini et al., 2006). Typically, thematic maps are derived from both classification of remotely sensed images and from data analysis in geographic information system (GIS) technology (Gopal and Woodcock, 1994). In a classical land cover map, a polygon or a pixel can describe only a single land cover category applying a boolean membership in the integer set {0, 1}; thus, the degree to which it is in reality mixed cannot be differentiated (Rocchini and Ricotta, 2007). On the contrary, fuzzy set theory allows map producers to maintain uncertainty information of each class by taking into account the gradual change from class membership to non-membership (Gopal and Woodcock, 1994). The fuzzy membership function μ associates for each entity (a polygon or a pixel) a membership level μ_c in the range 0 to 1 expressing the possibility that a given entity (i.e. a polygon or a pixel) belongs to the thematic map class *c* (see Zadeh, 1978). Several studies have used fuzzy set theory for forest mapping (see e.g. Maselli et al., 1995; Zadnik Stirn, 2006) proving its efficiency in discriminating forest versus non-forest areas applying a continuity criterion which seems to better represent ecological gradients within a landscape. However, most of the European and Mediterranean forest monitoring and conservation programs (e.g. European Forest Monitoring Programme, Forest Focus, ICP forests) rely only on crisp classified land cover maps. In this paper, we consider European land cover projects such as the CORINE Land Cover 2000 (EEA 2005a, Neumann et al., 2007), the Global Land Cover (GLC2000, Bartholomé and Belward, 2005) and the JRC Forest/Non-forest (Pekkarinen et al., 2007), which seem to produce the mostly used data for both research and management of forest ecosystems.

The aim of this paper is to assess the amount of classification uncertainty related to crisp classes, particularly focusing on forest mapping uncertainty. In particular we will quantitatively assess (i) the separability of boolean classes with respect to forest fuzzy class membership (μ_f), and (ii) the amount of μ_f inner variability of each crisp land cover class.

Materials and methods

Study area

The study area is the Tuscany region (Italy, 42°-44° north latitude, 9°-12° east longitude, WGS84, 3,464 km^2, *Fig. 1*), a typical Mediterranean ecosystem, including forests at various altitudinal locations. Currently, the surface of the region is predominantly occupied by agricultural areas (55 % of the surface), while the wooded areas occupy 34 % of the territory. Besides conifer plantations which are scattered throughout the whole region, broadleaf forests vary from the evergreen Mediterranean forests dominated by *Quercus ilex*, along the coastlines, to the *Fagus sylvatica* and *Abies alba* forests of mountain sites (see e.g. Chiarucci and Bonini, 2005; Rocchini, 2007).

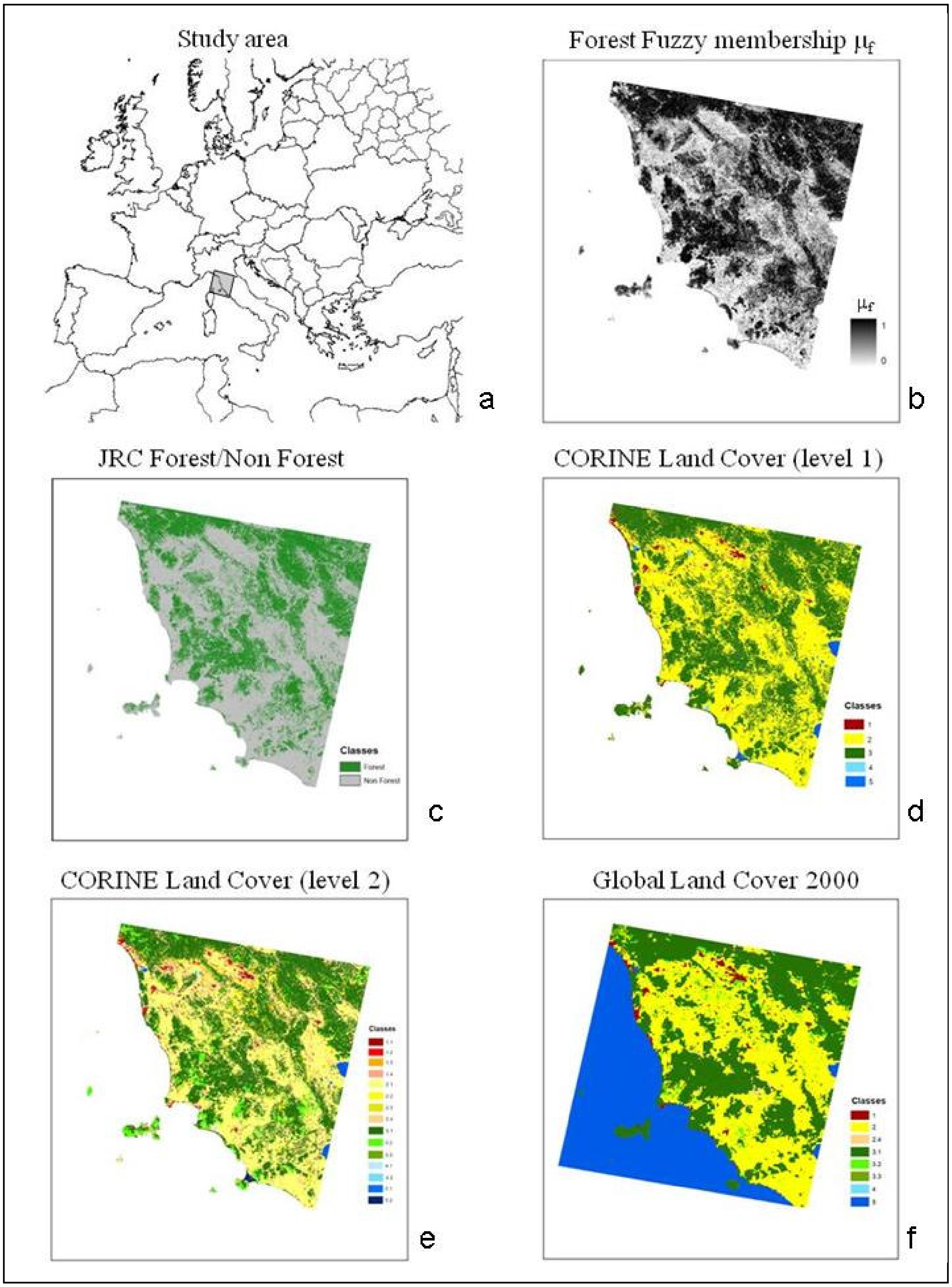

Figure 1. *Study area (a) and the used datasets:*
(b) Forest fuzzy membership μ_f*; (c) JRC Forest/Non-forest map; (d) CORINE Land Cover 2000 level 1; (e) CORINE Land Cover 2000 level 2; (f) GLC2000, rearranged according to the CORINE Land Cover dataset. See Table 1 for the classes description*

Landsat ETM+ data set and forest fuzzy classification

An ortho-rectified Landsat ETM+ image (path 192, row 030, acquisition date June 20, 2000, spatial resolution 30 metres in the optical/near infrared channels) was acquired. The bands used were: band 1 (blue, 0.45 – 0.515 µm), band 2 (green, 0.525 – 0.605 µm), band 3 (red, 0.63 – 0.69 µm), band 4 (near infrared, 0.75 – 9.90 µm), band 5 (middle infrared, 1.55 – 1.75 µm) and band 7 (middleinfrared, 2.09 – 2.35 µm). Band 6 was not considered due to the much larger pixel size than the other bands (60 meters of ground resolution opposed to 30 meters of the other bands).

In order to perform forest fuzzy classification, we applied a supervised classification approach by selecting known forest training sites (including both broadleaf and conifer plantations) within the image, based on field data acquired in previous research in Tuscany (Chiarucci et al., 2001; Chiarucci and Bonini, 2005; Rocchini, 2007).

A fuzzy soft classifier was applied using the software Idrisi (FUZCLASS module, Eastman, 2006). Fuzzy membership is based on the distance of each pixel to the mean reflectance of each band for a training class signature. To accommodate quality of training signatures and width of classes, the Z-score (standard deviation units) at which a fuzzy set membership decreases to zero was set to ~2 by trial and error procedure (Eastman, 2006). In this paper, we will refer to forest fuzzy membership to as μ_f.

European land cover crisp data sets

CORINE Land Cover 2000 dataset (levels 1 and 2)

The CORINE Land Cover project is a regional program in which the datasets have been created under the responsibility of each EU member state based on interpretation and digitisation of Landsat images (Landsat ETM+, with a resolution of 30 m) in a GIS environment (Bossard et al., 2000). The CORINE nomenclature comprises 44 land cover classes (at level 3), but in this paper we considered only levels 1 and 2 (5 and 15 classes, respectively; *Table 1*, *Fig. 1*). More specific levels like level 3 should allow to discriminate broadleaf from coniferous forests. However, as previously stated, the training sites for building the forest fuzzy map included both broadleaf forests and coniferous plantations, thus forcing us to choose higher CORINE hierarchical levels (i.e. a more generalised CORINE crisp classification) for performing further analysis (*Table 1*). We used the vector format dataset with a minimum mapping unit of 25 ha (EEA, 2005b).

Global Land Cover 2000 dataset (GLC2000)

The GLC2000 map (also developed by the JRC) covers the whole globe and uses the United Nations land cover classification system. It is based on Spot-Vegetation satellite images (with a spatial resolution of 1 km). This dataset was produced by harmonizing and merging the individual national products to one global product with a generalised legend (23 classes, see Bartholomé and Belward, 2005). In order to assure comparability, in this paper the codes of GLC2000 classes were rearranged according to the CORINE Land Cover classes scheme. Strictly spoken, the nearest CORINE class code was attributed to each GLC2000 class despite the CORINE level (*Table 1, Fig. 1*).

Table 1. *Crisp land cover classes for each dataset considered in this paper: CORINE Land Cover 2000 (levels 1 and 2), GLC2000, JRC. Notice that GLC2000 classes were rearranged according to the CORINE Land Cover dataset for comparability purposes. The number of points falling within each class is reported (total number of pixels = 6706).*

Dataset	**Code numbers and names of land cover classes**	**Number of points**
CLC2000 (level 1)	1 – Artificial Surfaces	232
	2 – Agricultural areas	3405
	3 – Forest and semi-natural areas	2996
	4 – Wetlands	10
	5 – Water Bodies	63
CLC2000 (level 2)	1.1 – Urban Fabric	154
	1.2 – Industrial, commercial and transport units	49
	1.3 – Mines, dumps and construction sites	20
	1.4 – Artificial non agricultural vegetated areas	9
	2.1 – Arable land	1831
	2.2 – Permanent crops	377
	2.3 – Pastures	221
	2.4 – Heterogeneous agricultural areas	976
	3.1 – Forest	2657
	3.2 – Scrub and/or herbaceous vegetation associations	322
	3.3 – Open spaces with little or no vegetation	17
	4.1 – Inland wetlands	9
	4.2 – Coastal wetlands	1
	5.1 – Continental waters	53
	5.2 – Marine waters	10
GLC2000	1 – Artificial surfaces and associated areas	123
	2 – Cultivated and managed areas	2994
	2.4 – Mosaic: Cropland/Shrub and/or grass cover	61
	3.1 – Tree Cover, broadleaved, deciduous, closed	1705
	3.1 – Tree Cover, mixed leaf type	778
	3.1 – Tree Cover, needle-leaved, evergreen	81
	3.2 – Herbaceous Cover, closed-open	282
	3.2 – Shrub Cover, closed-open, deciduous	354
	3.2 – Shrub Cover, closed-open, evergreen	211
	3.3 – Bare areas	43
	4 – Regularly flooded shrub and/or herbaceous cover	2
	5 – Water bodies	72
JRC Forest	Forest	2562
	Non forest	4144

JRC Forest/Non-forest dataset

The Pan-European Forest/Non-forest map (Pekkarinen et al., 2007) developed by the Joint Research Centre (JRC, *Fig. 1*) is based on Landsat ETM+ imagery with original spatial resolution of 30 m, subsequently resampled to 25 m with the scope of representing the forested area extent of the year 2000. The land cover information used for the forest mapping originates from the CORINE Land Cover 2000 (hereafter simply referred to as CORINE Land Cover). The categories constituting this map, related to main land cover classes are: 1-Forest, 2-Non Forest, 3-Clouds or snow (no data).

All the forest classes are defined as areas occupied by forest and woodlands with a vegetation pattern composed by native or exotic coniferous and/or broad-leaved trees. We refer to the JRC Pan-European Forest Mapping website (http://forest.jrc.it/ForestResources) for a complete description of those vegetation types included into the forest class.

Statistical analysis: testing crisp classes separability

Ten thousand random points were selected within the Landsat ETM+ image extent. Of these, 3294 points falling within the sea were not considered since the interest was focused on terrestrial ecosystems. Forest fuzzy membership μ_f values were attributed to each of the 6706 random points. The random points were labelled with each crisp class of each land cover dataset: JRC, CORINE Land Cover 2000 (levels 1 and 2), and GLC2000. Thus, a box-plot of forest fuzzy membership μ_f vs. classes for each land cover dataset was built in order to qualitatively assess the separability of crisp classes with respect to μ_f. The whole procedure was repeated considering different land cover classifications (CORINE Land Cover 2000, GLC2000, JRC,) and thematic levels. In this paper, we refer to thematic level as the hierarchical classification level of CORINE Land Cover data set.

In order to quantitatively test for μ_f separability of crisp classes considering each land cover dataset, a Kruskal-Wallis non-parametric test (Zar, 1996) was performed within the R statistical language software (R Development Core Team, 2007). The H_0 hypothesis to be tested was that the mean of forest fuzzy membership values is the same in each group (class) and is tested by a rank sum test. We refer to Rogerson (2001) for an analytical dissertation on the matter.

Results and discussion

Considering the CORINE Land Cover level 1, the land cover classes were statistically different ($p<0.001$) with respect to μ_f (*Table 2*); particularly class 3 (“Forest and semi-natural areas”) showed higher μ_f values (*Fig. 2*).

Table 2. *Separability of crisp land cover classes with respect to forest fuzzy membership tested by Kruskal-Wallis non-parametric test.*

Dataset	Chi-squared	df	p-value
CORINE Land Cover level 1	2767.85	4	<0.001
CORINE Land Cover level 2	2969.31	14	<0.001
GLC2000	1621.47	7	<0.001
JRC	2829.99	1	<0.001

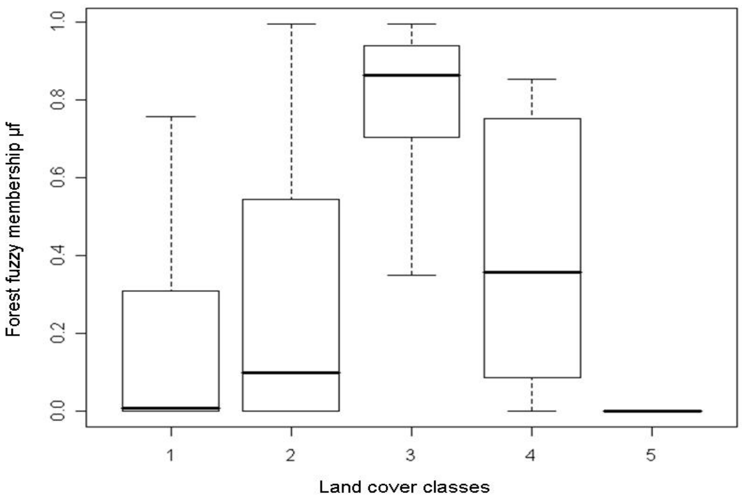

Figure 2. *Box-plot of forest fuzzy membership μ_f versus CORINE Land Cover classes (level 1). See Table 1 for the classes description and the legend of Fig. 2 for the description of box-plot graphical representation.*

High μ_f values were detected for wetland areas (class 4) where shrublands and riparian tree vegetation (dominated by *Populus* and *Salix* genera) should have led to an increased μ_f value (considering both median and variability towards higher μ_f values). Variability in lower μ_f values is presumably related to the presence of shallow water. We refer to Rocchini et al. (2005) and Laba et al. (2008) for major details on spectral behaviour of wetland vegetation and its classification.

Agricultural areas (class 2) showed low values approaching 0.1 but with a high variability reaching maximum μ_f values mainly related to tree crops (fruit, olive plantations, etc). Urban areas (class 1) and water (class 5) showed μ_f values approaching zero, with a high and very low variability, respectively. The high μ_f variability of urban areas is probably due to urban parks and isolated trees (La Sorte and McKinney, 2006; Ricotta et al., 2001).

The same general pattern was found for the CORINE Land Cover level 2 (*Fig. 3*) with classes being statistically different ($p < 0.001$) to each other (*Table 2*). For a major comprehension of the achieved results (*Fig. 3*), we will discuss CORINE Land Cover level 2 from *urban areas* (classes starting with code 1) to *water* (classes starting with code 5).

Urban fabric and *industrial units* (classes 1.1 and 1.2) showed low median values. Surprisingly, both *mines* and *artificial non agricultural vegetated areas* (1.3, 1.4) showed higher μ_f values approaching 0.3 with a high variability. Nonetheless, it should be stressed that the number of points falling within such classes was quite low (20 and 9 respectively, *Table 1*). Thus it is hypothesised that the low number of points should hamper to completely evaluate actual μ_f median and variability values. Considering *arable land*, all classes (2.1- *Arable land*, 2.2 - *Permanent crops*, 2.3 - *Pastures*, 2.4 - *Heterogeneous agricultural areas*) showed low μ_f values ranging from ca. 0.03 to 0.2 with variability being probably dependent of orchard areas.

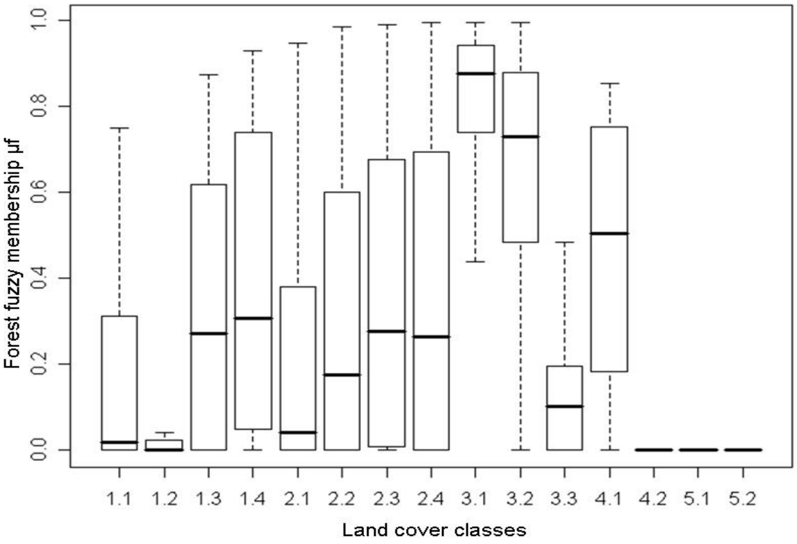

Figure 3. *Box-plot of forest fuzzy membership μ_f versus CORINE Land Cover classes (level 2). See Table 1 for the classes description and the legend of Fig. 2 for the description of box-plot graphical representation.*

Once *forests and semi-natural areas* (level 1, class 3) were divided into more detailed classes, *forests* (class 3.1) showed highest μ_f values. Meanwhile, considering their variability, low values were achieved approaching the value 0.4. *Scrub and/or herbaceous vegetation associations* (class 3.2) showed intermediate median values with a high variability spanning the whole μ_f range, while open spaces (class 3.3) showed low median values with variability standing at the low μ_f values. As previously stated for *wetlands* (level 1, class 4) inland wetlands (4.1) showed median values approaching 0.5 and a high variability mainly due to riparian vegetation. On the contrary, *coastal wetlands* (4.2) showed a low value related to a single point falling within this class (*Table 1*).

Water classes, both continental (class 5.1) and marine (class 5.2), showed μ_f values approaching 0 with very low variability as found considering the CORINE Land Cover level 1.

Considering the GLC2000 dataset rearranged with the CORINE Land Cover level 1 and 2 legend, while the same general pattern of the CORINE Land Cover level 1 and 2 was observed (*Table 2, Fig. 4*), some differences were detected. As an example, *wetland areas* (class 4) showed μ_f values higher than those of *forests* (class 3.1). However, it should be noticed that class 4 was represented by only regularly flooded shrub and/or herbaceous cover thus increasing μ_f median values, and only two points fell within this class (*Table 1*). The forest map result of fuzzy classification performed through the selected 6 Landsat ETM+ bands used is shown in *Fig. 1*.

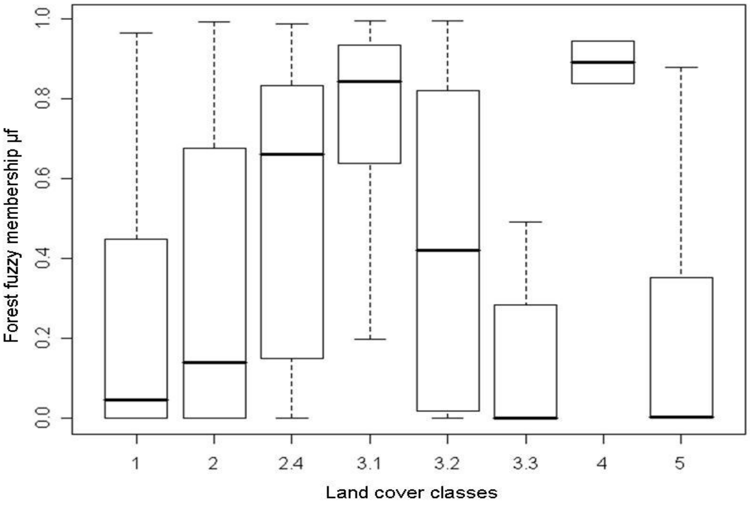

Figure 4. *Box-plot of forest fuzzy membership μ_f versus GLC2000 classes. Notice that codes were assigned according to the CORINE Land Cover thematic legend. See Table 1 for the classes description and the legend of Fig. 2 for the description of box-plot graphical representation.*

The JRC dataset showed a statistically significant separability ($p < 0.001$) of forest fuzzy membership μ_f between forest versus non-forest classes (*Table 2*), with forest class showing higher values with respect to non-forest class, as expected (*Fig. 5*). The lower μ_f values variability of forests should be related to substantial landscape homogeneity of this class within the Tuscany region.

In this paper, fuzzy classification was applied only to forest mapping (i.e. forest fuzzy membership) in order to assess the amount of variability which is lost by using crisp sets for forests identification within a landscape. We focused on only one Landsat ETM+ image scene within a typical Mediterranean area such as the Tuscany region, on the strength of its high landscape variability (Rocchini, 2007), in order to stress the whole analysis on a high level of uncertainty in forest identification to deal with. However, this should not hamper to translate the achieved results to other types of landscapes or thematic classes.

The considered crisp datasets showed statistically significant μ_f separability. However, despite the hierarchical level being taken into account, both forest and non-forest crisp classes showed a high degree of inner μ_f variability (uncertainty). Focusing on forest classes, μ_f reached in some cases low (~0.4, see e.g. *Fig. 3 and 4*) to very low values (~0.2, see e.g. *Fig. 5*). Two main reasons could be advocated for: (i) scale matching problems associated with different spatial resolutions (Jelinski and Wu, 1996), (ii) the intrinsic mixing of crisp land cover classes (Small, 2004).

As stressed by Arbia et al. (1998), one of the main problems when dealing with digital representation of geographic entities is linked to the discrepancy between the digital GIS representation of spatial objects and the real entities they are seeking to describe. In particular, it is worth recalling that results of the analyses for the same area can vary because of the spatial resolution (Johnson and Howarth, 1987), and that some patterns or processes can be recognised only at appropriate resolution (Jelinski and Wu, 1996). An ecological phenomenon or a spatial entity may remain undetected because of an improper matching with the scale of analysis (Stohlgren et al., 1997). Therefore, scale mismatching may hamper the comparison of spatial entities with different resolutions. As an example, it is expected that a polygon layer with a minimum mapping unit of 25 ha (as that of the CORINE Land Cover) or a raster layer with a pixel resolution of 1 km (such as that of the GLC2000) should thematically match the Landsat ETM+ pixel of 30m.

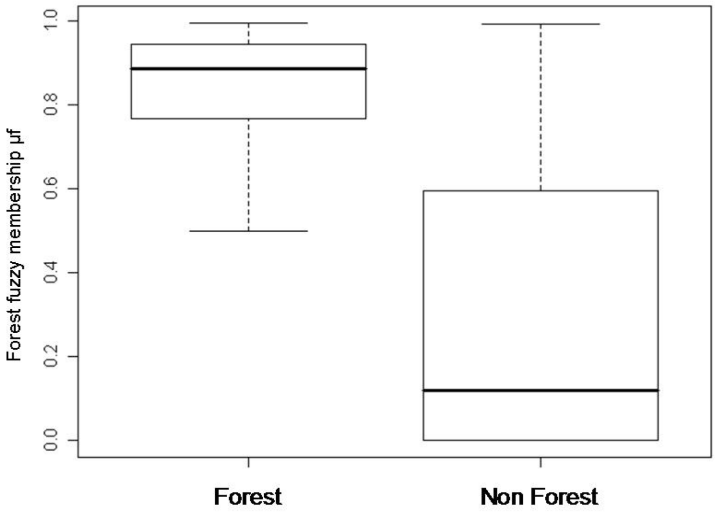

Figure 5. *Box-plot of forest fuzzy membership* μ_f *versus JRC Forest/Non-forest classes. The box represents the middle 50% of the data, the upper and lower parts of the box indicate the 75th and 25th percentile of the data, respectively. The line in the box indicates the median value of the data and the length of the bars indicate 1.5 times the inter-quartile distance.*

Spatial resolution has impact on the thematic resolution of the represented entities (Burrough and McDonnell, 1998; Nagendra and Rocchini, 2008). In fact, more detailed crisp classes can be represented only at higher spatial resolution with smaller spatial objects (e.g. hedgerows), while a higher degree of spectral mixing and inner thematic uncertainty is expected when objects (both polygons and pixels) become bigger.

Conclusion

In this paper, we demonstrated that crisp classification which is being continuously used in landscape research and planning is not free from drawbacks as it is showing a high degree of inner variability despite even detailed classification schemes and hierarchical levels being considered. We are not claiming at dismissing crisp classification, but we suggest taking problems into account which derive from a crisp view of ecological entities being mapped. Since habitats are expected to gently and continuously vary rather than abruptly change within a landscape.

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THE RELATIONSHIP BETWEEN LUNAR PHASES AND THE EMERGENCE OF THE ADULT BROOD OF INSECTS

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Abstract. Based on many years of light trap catch data by the Hungarian Light Trap Network and other traps, we have examined the relationship between the emergence of the adult broods of insects and lunar phases. The summarized results of adult brood emergences were plotted against 30 phase angle divisions, our special dividing method of the lunation also used in our previous works. We have found that the emergence of the adult brood of several Macrolepidoptera and Coleoptera species is associated with a special lunar phase, most often with the Last Quarter.

Keywords: *insects, moon phases, first specimen of generation*

Introduction

A reliable prediction of the adult brood emergence of pest insect species is of extreme importance in plant protection. Environmentally sound, effective and economical protection treatments are usually timed to correspond with a special phenological period of the pest species, which can generally be calculated with acceptable precision from the starting date of the adult brood.

By “adult brood” or “flight” we mean the length of the flight-period of a given adult generation of a species. The length of flight is usually longer than the lifespan of a single adult specimen. Adults are usually short-lived, thus flight can be considered as a time series of the difference of emerging and dying individuals. Accordingly, the aspect of flight reflects – at least partially – the hatching of imagos from the pupae, with some probably negligible delay. This aspect may also be modified by the number of individuals arriving to and departing from the given location.

The phenological phenomena of insects are connected to meteorological factors, including primarily temperature. The rate of the metabolic processes of poikilotherm insects is determined by the temperature of the environment. As a result of observations and experiments, scientists found that the beginning of the flight of numerous species correlate with the sum of heat units required for development. (Nowinszky 2004; Nowinszky et al., 2007, 2008)

Review of literature

In the view of several authors, the life cycle and thus the emergence of the adult brood of insects is possibly also in relation with lunar phases. Research in this field has mainly concentrated on short-lived species like Ephemeroptera, where the beginning and the culmination of the flight occur in quick succession. Hora (1927) compared old records on some Ephemeroptera – the first one from 1744! – with the lunar phases and found that the adults emerged most frequently in the vicinity of a Full Moon. Pongrácz (1933) experienced several times that the adults of the mayfly species *Polymitarcis virgo* Oliv. emerged en masse right after a Full Moon. According to Csongor and Móczár (1954), the swarming of *Palingenia longicaudata* Oliv. (Ephemeroptera) is affected by rising air pressure as well as rising air and water temperatures and the lunar phases. After an examination of the records of the greater swarms of mayflies, Móczár (1957) pointed out that they had all occurred one day after a Full Moon or a New Moon. Between March 1953 and April 1955, Hartland-Rowe (1955) observed that the 22 swarms of *Povilla adusta* Naval (Ephemeroptera) at Lake Victoria and Lake Albert happened in a 5-day vicinity of – most often 2 days after – a Full Moon. Adults only live for about one hour. Between 10th and 21st August 1954, 1521 specimens were collected by light trap, 44% arriving on the second day after a Full Moon. The author proposes this lunar rhythm only to be true near the Equator where day length and temperature are quite stable throughout the year. Also at Lake Victoria, MacDonald (1956) found that the swarming of some unidentified Chironomus and Chaoborus species occurred shortly after a New Moon. The author takes the influence of the Moon on the biology of these species as evidence. For a further examination of this question, Corbet (1958) by use of Robinson-type light traps equipped with 125 W mercury vapour bulbs, collected Plecoptera, Ephemeroptera, Chaoborus and Trichoptera species on one hundred consecutive nights on the shore of Lake Victoria. In four of the 37 species examined, the number of individuals showed a periodical change corresponding to the change of lunar phases. This phenomenon could not be proved for the rest of the species. In his view, the timing of the highest and lowest catch in the vicinity a Full or a New Moon does not confirm the assumption that the light of the Moon has an influence on the light trap catch. The reason might rather be the lunar rhythm of adult emergence. Relying upon their suction trap catch results, Perfect and Cook (1982) presume a synchronization mechanism between lunar phases and the flight activity of the species living in the tropics where the change of temperature is moderate. According to Fryer (1959), the rhythm of the development of the non-biting midge species *Chironomus brevibucca* Kief. follows Moon Phases closely. Adults emerge in great numbers after a Full Moon. On the other hand, Ito et al. (1993) did not observe a synchronicity between lunar phases and the emergence of *Scotinophara coarctata* (Thunberg) (Heteroptera: Pentatomidae).

It is important to clearly separate lunar influence on adult brood emergence and one on activity. Fryer (1959) also notes that in the case of *Chironomus brevibucca* Kief, the two periods inevitably coincide. Although it is not sure that lunar influence also affects the beginning of the flight of longer-lived insects. Numerous authors share the view that lunar phases might synchronize the biological rhythm of Lepidoptera. Persson (1974) observed an unfavourable influence of moonlight on the oviposition of *Heliothis zea* Boddie. According to a presumption by Nemec (1971), the generation cycles of *Heliothis zea* Boddie are synchronized with Moon Phases. The results of Agee et al. (1972) also suggest the activity and life cycle of *Heliothis zea* Boddie to be synchronized with lunar phases. Possibly, several other physical phenomena – like the maturity of the foodplant, temperature or other factors – might also be responsible for this lunar rhythm.

Field observations indicated that the peak of the emergence and mating of tobacco budworm (*Heliothis virescens* Fabr.) occurred 8-9 days before the capture-peaks in pheromone traps. In addition, mating peaks occurred 4 days before a Full Moon while trap capture peaks 4 days after a Full Moon (Raulston et al., 1979).

Bowden and Gibbs (1973) find it worth considering whether lunar phases synchronize flight activity in some stage of development. According to Bowden (1973a), lunation determines an important biological rhythm for the different insect species. The conclusion of Danthanarayana (1986), that Moon periodicity is unrelated to the time of the appearance of the Moon above the horizon is of great importance. This precludes the possibility that the observed flight behaviour would be a direct visual response to the Moon or to moonlight. In his view, the periodicity of the Moon may have the advantage of making survival possible via migration and contributing to the synchronization of emergence and flight in favour of mating and reproduction.

Roux and Baumgartner (1995) operated pheromone traps on Tunisian potato fields between 1986 and 1991 to record the number of male specimens of *Phthorimaea operculella* (Zeller) weekly. A four-week cycle – presumably influenced by the Moon – was identified.

Conlee (1995) analyzed 10 years of daily sex pheromone-baited trap capture data for the codling moth (*Cydia pomonella* L.) for lunar periodicity, using both statistical and non statistical methods in South-California. Using time series (spectral) analysis, a sine wave periodicity of approximately 30 days was detected in 6 of 10 annual spectra and at a significance level of $P < 0.05$ in a combined 10-year spectrum. Auto regression analysis failed to detect precise periodicity and showed that these trap captures are independent after 3 days. Frequency distributions of generation emergences over 10 years showed that 80% occurred within either 3 days of a New or a Full Moon. First true generation emergence was found to change according to coincidence of the lunar periodicity to annual constraints, and second generation emergence occurred 43.4 days (1.5 lunar cycles) later. A hypothetical model for this was developed and its potential accuracy compared with that of published degree-day models. This model has accurately predicted first and second generation codling moth emergences in southern California during the past two years.

Materials and methods

For our research, we used the complete Macrolepidoptera material of the forestry light trap operated in the Kámon Botanical Garden in Szombathely between 1962 and 1970, including data of 3395 adult broods. We have considered the catch of the first specimen of a species as the beginning of the flight. In the case of polyvoltine species, we only considered the first generation, if the generations were not clearly separable.

We have also processed data provided by the 49 observation posts of the National Agricultural and Forestry Light Trap Network on 31 species represented by more than 20 specimens from each generation (Nowinszky and Ekk, 1988). Most of our catch data come from the period 1959-1970, but we also utilized earlier data from the light trap network being developed steadily over the years as from 1953. We have processed the catch results of several important pest species until 1976. We have also worked with data on the winter moth *Operophtera brumata* L. for the period 1961-1976, including 203 adult broods. We were also provided with the catch data of the National Light Trap Network on the 245 and 151 adult broods of fall webworm *Hyphantria cunea* Drury for the periods 1953-1963 and 2000-2006, as well as on the 673 broods of the Common Cockchafer *Melolontha melolontha* L. for the period 1966-2006.

We also used all data of the Macrolepidoptera caught by the special light trap operated by Varga and Mészáros (1973) in Becse (Voivodina, Serbia) between 1970 and 1973, and the Coleoptera data of the same trap collected from 1971.

We have downloaded the phase angle values of the Moon from the following URL: http://aa.usno.navy.mil/data.docs/MoonPhase.html. We have divided the 360° phase angle of the full lunar month (lunation) into 30 divisions. The division in the ± 6° phase angle value vicinity of a Full Moon (0°, or 360°) was named: 0. Starting from here, divisions in the direction of the First Quarter until the New Moon were named: -1, -2, -3, -4, -5, -6, -7, -8, -9, -10, -11, -12, -13, -14. Also starting from the Full Moon, divisions in the direction of the Last Quarter until a New Moon were named: 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, and 14. The division including the New Moon was named: ±15. All divisions include 12 phase angle values. We have arranged all nights of the observation period into one of these phase angle divisions (Nowinszky, 2003).

Consequently, we have assigned the day of the catch of the first specimen of all adult broods of all species to a phase angle division of the Moon, summarized the results, and finally we plotted in percentage the frequency of adult brood emergences falling into the different divisions.

Results

The beginning of the flight of the Macrolepidoptera species caught by the light trap working in Kámon Botanical Garden plotted against lunar phases is presented in *Fig. 1.*

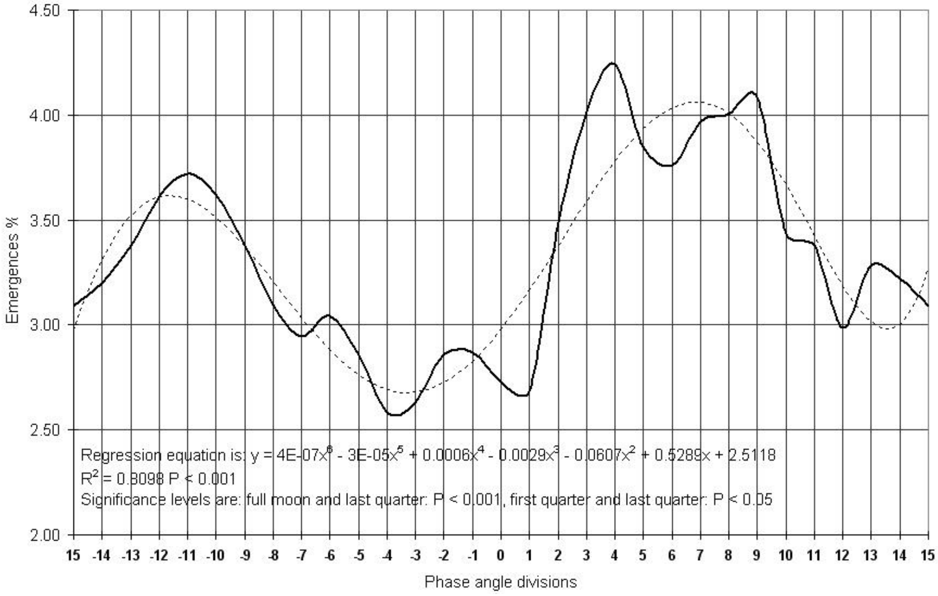


Figure 1. *The emergence of the adult broods of the processed 31 Macrolepidoptera species caught by the light-trap working in Kámon Botanical Garden (1962-1970, 556 species)*

The emergence of the adult broods of the processed 31 Macrolepidoptera species caught by the National Light Trap Network is illustrated in *Fig. 2-6. Fig. 2* shows the summarized curve of the five species beginning to fly vicinity a Full Moon. *Fig. 3* includes the summarized curve of the species starting to fly in the vicinity the Last Quarter. The curves of 2 species starting to fly at a New Moon and after a Full Moon are illustrated in *Fig. 4. Fig. 5* shows the curve of species starting to fly in the period of a New Moon and in the Last Quarter.

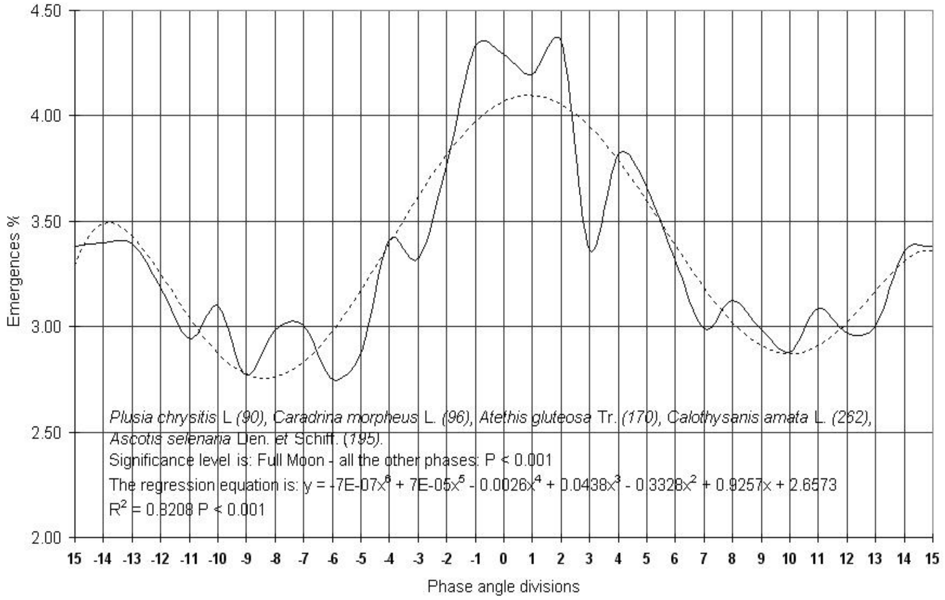


Figure 2. *The emergence of the adult broods of the processed 5 Macrolepidoptera species caught by the National Light Trap Network. The summarized curve of the species beginning to fly in the vicinity of a Full Moon.*

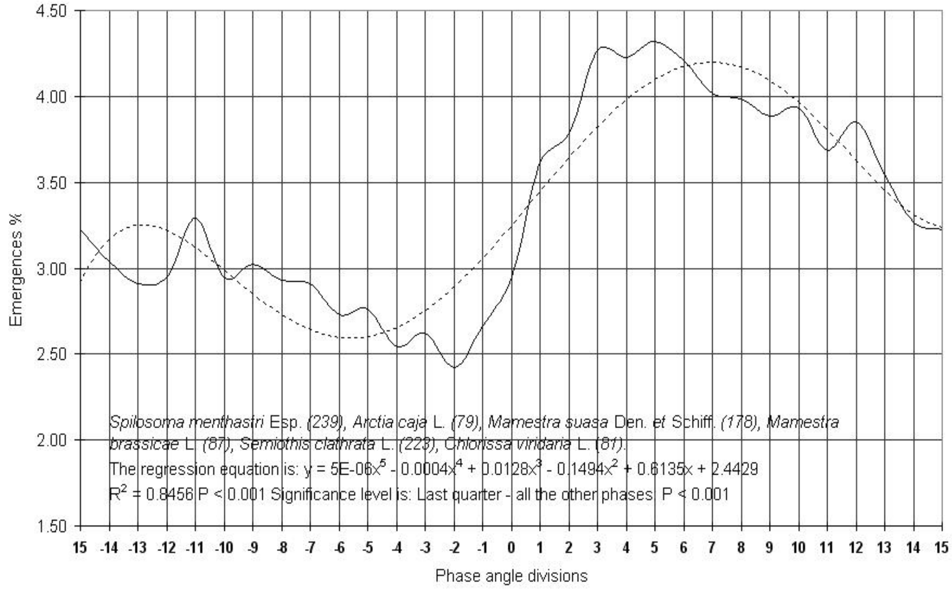


Figure 3. *The emergence of the adult broods of the processed 6 Macrolepidoptera species caught by the National Light Trap Network. The summarized curve of the species beginning to fly in the vicinity of a Last Quarter*

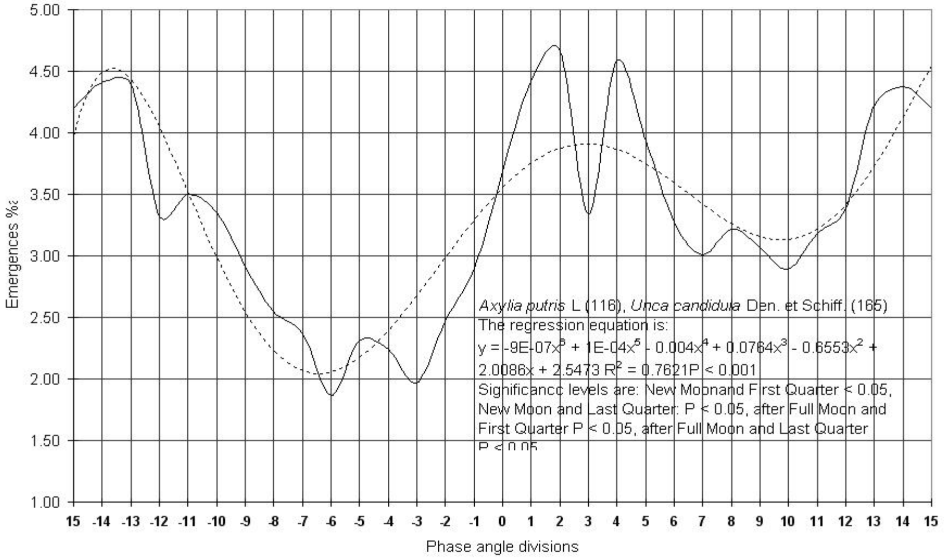


Figure 4. *The emergence of the adult broods of the processed 2 Macrolepidoptera species caught by the National Light Trap Network. The summarized curve of the species beginning to fly in the vicinity of New Moon and after Full Moon*

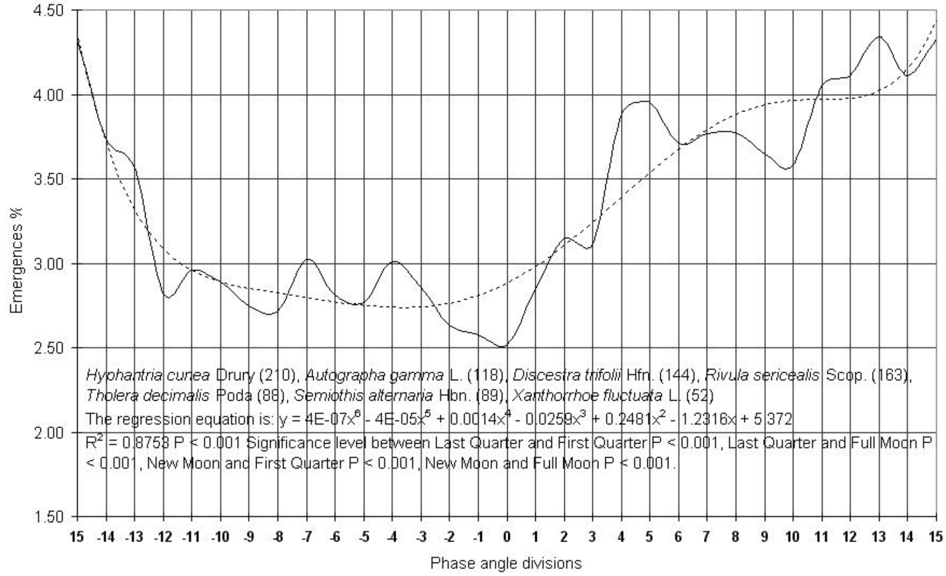


Figure 5. *The emergence of the adult broods of the processed 7 Macrolepidoptera species caught by the National Light Trap Network. The summarized curve of the species beginning to fly in the vicinity of New Moon and Last Quarter*

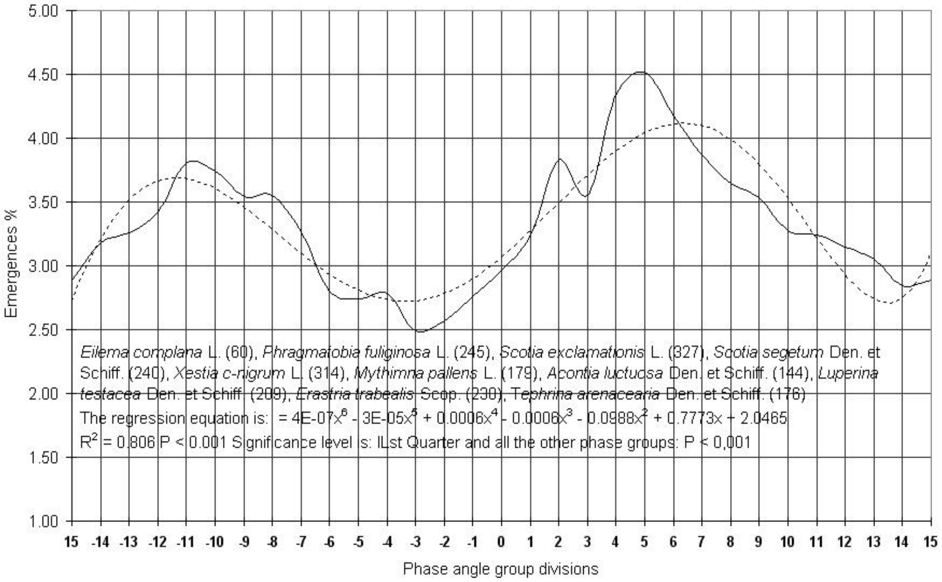


Figure 6. *The emergence of the adult broods of the processed 10 Macrolepidoptera species caught by the National Light Trap Network. The summarized curve of the species beginning to fly in the vicinity of First quarter and Last Quarter*

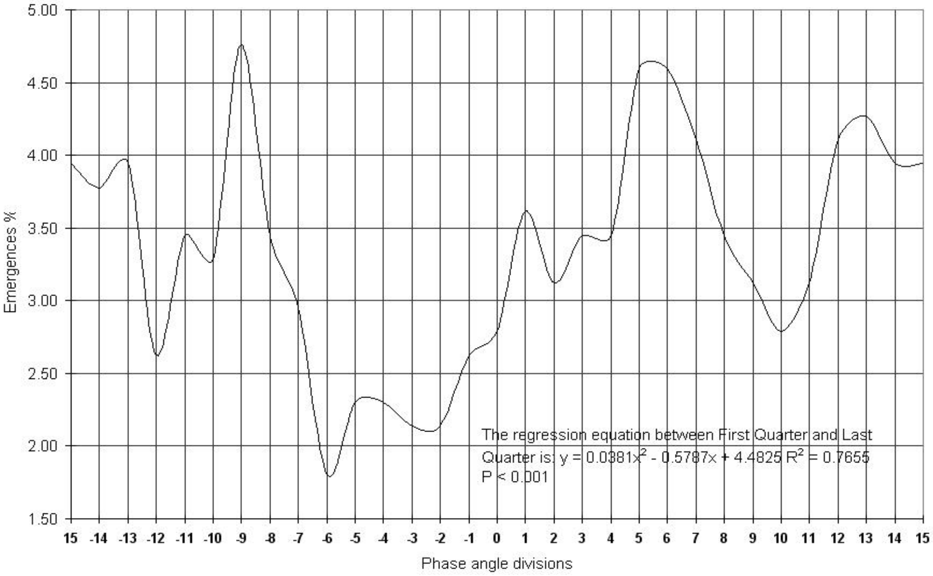
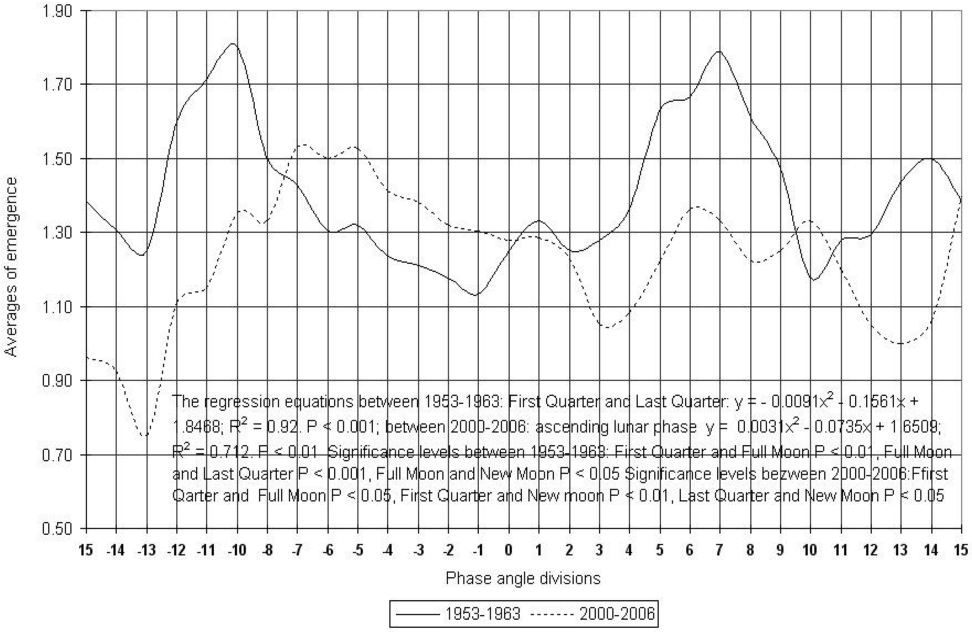


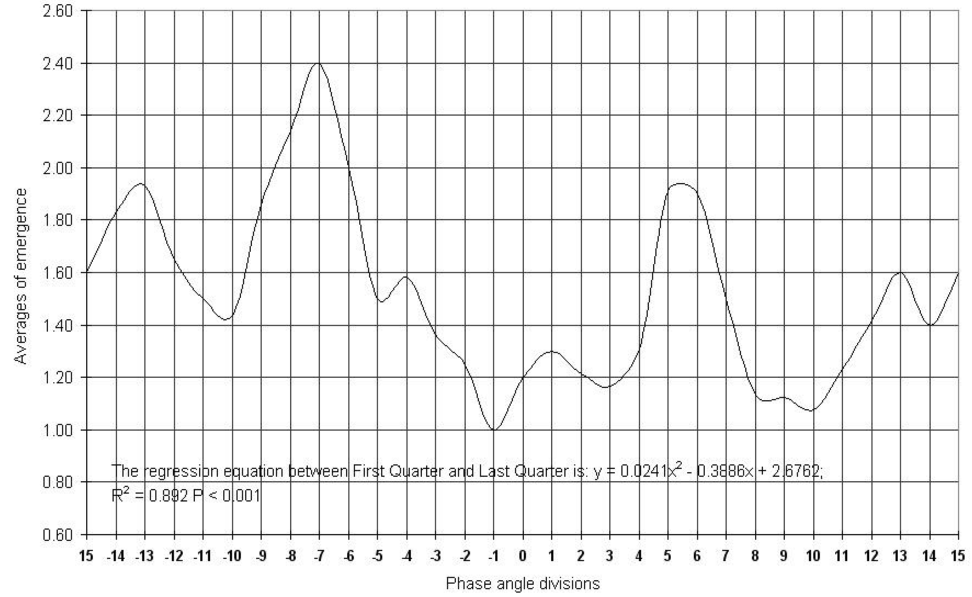
Figure 7. *The emergence of the adult broods of winter moth (Operophtera brumata L.), depending on the lunar phases from the material of the National Light Trap Network (1961-1976)*

In *Fig. 6* species beginning to fly in the First and Last Quarters can be seen. It was in the case of a single species (*Spilosoma lubricipeda* Esp.) alone where no connection between the lunar phases and the beginning of flights could be proved. *Fig. 7* shows the emergence of the adult broods of the Winter Moth *Operophtera brumata* L. plotted against lunar phases. The emergence of the adult broods of the Fall Webworm (*Hyphantria cunea* Drury) between 1953-1963 and 2000-2006 is illustrated in *Fig. 8. Fig. 9* shows the beginning of 673 adult broods of the Common Cockchafer (*Melolontha melolontha* L.).



The regression equations between 1953-1963: First Quarter and Last Quarter: $y = -0.0091x^2 - 0.1561x + 1.8468$; $R^2 = 0.92$. $P < 0.001$; between 2000-2006: ascending lunar phase $y = 0.0031x^2 - 0.0735x + 1.6509$; $R^2 = 0.712$. $P < 0.01$ Significance levels between 1953-1963: First Quarter and Full Moon $P < 0.01$, Full Moon and Last Quarter $P < 0.001$, Full Moon and New Moon $P < 0.05$ Significance levels bezween 2000-2006:Ffirst Qarter and Full Moon $P < 0.05$, First Quarter and New moon $P < 0.01$, Last Quarter and New Moon $P < 0.05$

Figure 8. *The emergence of the adult broods of fall webworm moth (Hyphantria cunea Drury), depending on the lunar phases from the material of the National Light Trap Network (1953-1963 and 2000-2006)*



The regression equation between First Quarter and Last Quarter is: $y = 0.0241x^2 - 0.3886x + 2.6762$; $R^2 = 0.892$ $P < 0.001$

Figure 9. *The emergence of the adult broods of common cockchafer (Melolontha melolontha L.), depending on the lunar phases from the material of the National Light Trap Network (1966-2006)*

The emergence of the adult broods of the processed 73 Macrolepidoptera and 93 Coleoptera species caught by the light trap of Varga and Mészáros are illustrated in *Fig. 10* and *Fig. 11*.

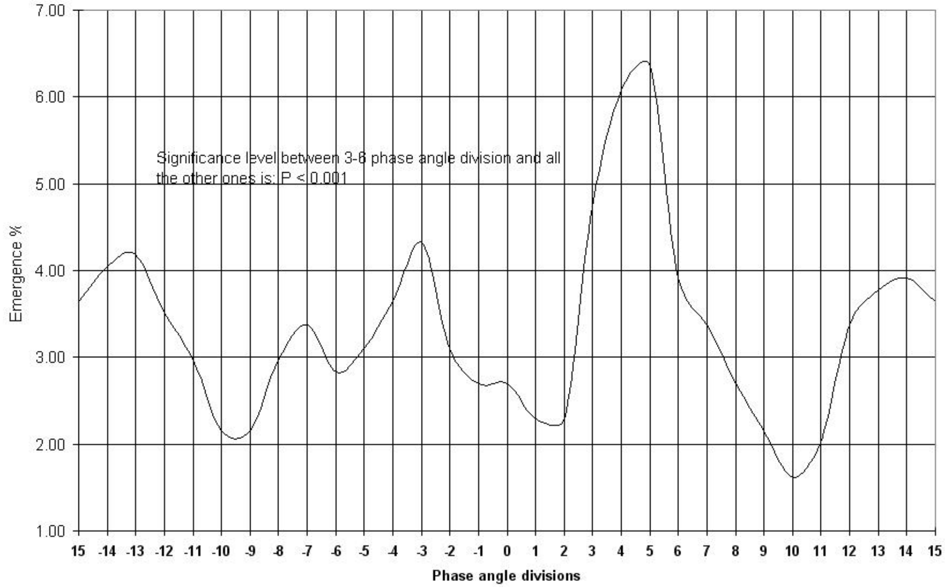


Figure 10. *Emergence of Macrolepidoptera species caught by the Varga-Mészáros type light-trap in Becse (Voivodina, Serbia)*

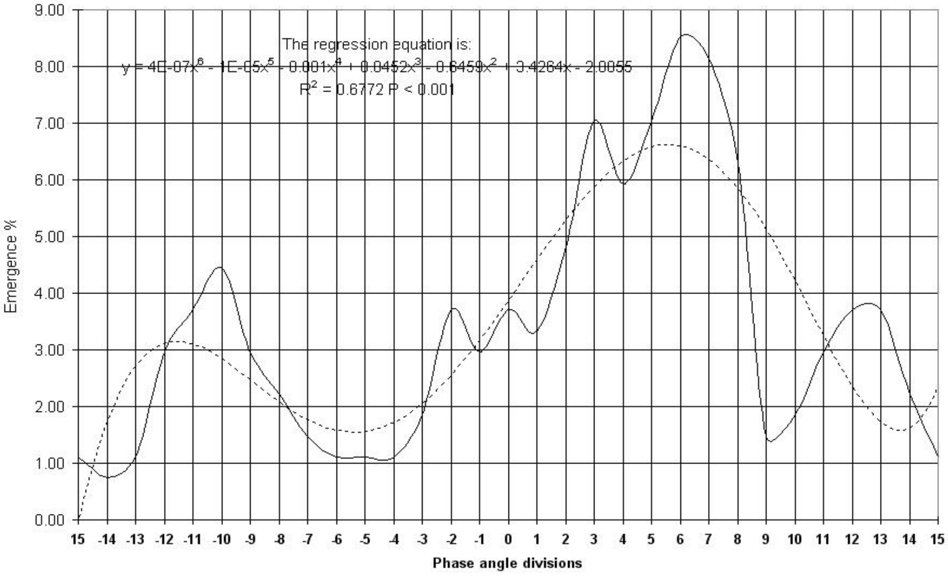


Figure 11. *The emergence of the Coleoptera species caught by the Varga-Mészáros type light-trap in Becse (Voivodina, Serbia)*

Discussion

The examination of the Macrolepidoptera material of the National Light Trap Network leads to results very similar as the ones based on data of the Kámon Botanical Garden (*Fig. 1*). The flight of 23 species was proved to start most often in the last quarter, or also in the last quarter. The flight of only five species began most often at a full moon. The highest frequency values of the adult brood emergences of the species examined from the material of the National Light Trap Network can all be well described by quadratic polynomials.

The flight of the Winter Moth (*Operophtera brumata* L.) is significantly more frequent in the waning than in the waxing phases of lunation. There is also a significant difference between the frequencies of the First and Last Quarter, the ones of a Full Moon and the Last Quarter and the ones of a Full Moon and a New Moon. The frequency of flights beginning between phase angle divisions -9 and 5 can be described by a quadratic polynomial. It is remarkable that the increase lasts only until phase angle division 5. An explanation for the latter might be that in the vicinity the Last Quarter, the Moon is above the horizon only after midnight, the approximate time until the species could experience the presence of the Moon.

The beginning of the flights of the Fall Webworm (*Hyphantria cunea* Drury) between 1953-1963 and 2000-2006 do not show a significant deviation in any of the lunar quarters. Between 1953 and 1963, flights starting at a Full Moon are significantly less frequent than in the other lunar quarters. In the 2000-2006 period, this significant deviation is only true when compared to the First Quarter, but even then, flights start most often in the First and Last Quarters, like in the years 1953-1963. The broods of the Common Cockchafer (*Melolontha melolontha* L.) start most often in the First or Last Quarter. On the other hand, broods begin significantly more often in the waxing, than in the waning lunar phases. Adult broods emerge vicinity a Full Moon less frequently than in any other quarter.

Our results prove that the emergence of the adult brood of at least some of the Macrolepidoptera and Coleoptera species is associated with a special lunar phase, most often with the Last Quarter (*Fig. 3* and *Fig. 11*). The lifespan of Lepidoptera species is usually short, so there might be some kind of timing factor that ensures that mature individuals find each other. The regularly alternating lunar phases seem to be most suitable for this purpose. The decrease of light trap catch during a Full Moon reported by most scientists might, in our view, be at least partially explained by the relationship between the life cycle of insects and the moon phases.

Although we cannot preclude, that light traps cannot detect the emergence of adult broods in the same ratio during a Full Moon, as in the First and Last Quarters. This possibility raises the question of why adult broods emerge less frequently also around a New Moon, as moonlight might have no influence in these periods. Further investigation is certainly needed to answer this question.

On the other hand, it should also be noted, that the appearance of first specimen in the light trap does not necessarily mean the beginning of the flight of adults, though in the case of the common cockchafer (*Melolontha melolontha* L.) the trap catches specimen already on the very first day (Homonnay and Csehi, 1990). The first dispersing or migrating specimen caught by the light trap might cause further unreliability, as these might not have developed at the exact locality. This indicates a possible connection between migration and the lunar phases.

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INDICATION STRENGTH OF COENOLOGICAL SIMILARITY PATTERNS BASED ON GENUS-LEVEL TAXON LISTS

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Abstract. Several methods and indicators can be used to evaluate the coenological state of a given habitat, the ones which can be created simply, quickly, standardizably and reliably can be of outstanding practical importance in environmental biology. One possible method is listing the genera which can be found in a given habitat in great abundance and have little number of species and various ecological characteristics. For this purpose one of the most appropriate groups is that of ground-dwelling oribatid mites. In our research the indication strength of genus-level taxon lists and the effects of the main pattern-generating factors creating similarity patterns were analysed with the help of data series on oribatid mites collected by us and originating from literature. Our aim was to develop a method with the help of which the difference expressed with distance functions between two oribatid mite genus lists originating from any sources can correspond to spatial and temporal scales. With the help of this base of comparison changes in disturbed or transformed habitats can be expressed by means of oribatid mite communities, with spatial and temporal distances.

Keywords: *oribatid mite, genus list, family list, distance function, indication, pattern generation*

Introduction

Nowadays there are several methods to describe the natural state of a habitat; the focus is mainly on the measuring of biodiversity. However, uncertainty can arise when measuring biodiversity, as several questions can be raised already as to the explanation of the term, starting from which level it should be considered on (genetic, taxon, ecological diversity), to – if the taxon level has been chosen – the decision on which taxon the focus should be.

The main goal of this study is to set up a comparison scale based on genus-level presence-absence lists of oribatid mite communities (Acari: Oribatida) of habitats examined on different spatial and temporal scales. The secondary goal – and this time the precondition as well – is to get a reliable picture of the indication strength of the distances to be used, i.e. the information content included.

The indication suitability of the order of oribatid mites for describing the state of their habitat is justified by the special characteristics of the group. Oribatid mites can be found in almost all kinds of habitats: on land and in water; first of all in soil layers containing organic matter as well, however, they have penetrated into different other microhabitats, too (e.g. lichens, moss, bark etc.), which is mainly due to their indeed various food sources (e.g. organic debris, fungi, other mites etc.). Besides the diversity of habitats, their high adaptation ability is shown by their enormous abundance and species richness as well. The above characteristics can be mainly used in the case of coenological methods (Lebrun and van Straalen, 1995; Behan-Pelletier, 1999; Gulvik, 2007; Gergócs and Hufnagel, 2009).

The choice of the genus level can be explained by different aspects. In the analysis by Caruso and Migliorini (2006) it was shown that there were not any significant changes in data examining anthropogenic disturbance on oribatid mites when switching from species level to genus level. Our study has a similar goal as we would like to show potential habitat changes with our method. Podani (1989) had a similar observation in case of plants, according to which switching to genus level does not mean a significant change when comparing the examined habitats. Osler and Beattie (1999) carried out a meta-analysis similar to ours, which confirmed their expectation that taxonomic levels above species are more suitable for comparing habitats. This research showed further that habitats can be chosen on family level in case of oribatid mites, therefore our study covers besides the genus level the family level as well. There were also some other arguments for our decision, namely that the number of databases used could be considerably extended in this way, in addition, taxonomical processing became faster and more reliable in our field studies as well. Genus-level identification of oribatid mites is solved on the basis of the work by Balogh and Balogh (1992) on a global scale, too. However, species-level identification is only possible in case of some zoogeographical regions and only some taxa on a global scale as the related literature is not properly synthesized yet (e.g. Balogh and Mahunka, 1983; Olsanowski, 1996).

Main goals

By setting up the spatial and temporal scales, we expected that the order based on the genus lists and the family lists developed from them later should correspond to the real spatial and temporal scales, i.e. the farther and qualitatively the more different habitats our lists originate from, the greater difference there should be among similarities inside the given categories. However, if data originate from the same site, the difference among the examined samples should be greater in case of the lists which are farther in time from each other.

The main goals of the present study are the following:

1. Revealing the effect of the taxon level on data series based on species-, genus- and family-level lists.
2. Developing a spatial and temporal scales reference based on the genus- and family-level taxon lists with the help of similarity functions.
3. Examining the degree of distances in the similarity order used for indication.

Review of literature – suitability of oribatid mites as indicators

Research into oribatid mites goes back to the 1880s, the work of A. Berlese, who invented the Berlese funnel and made it possible to extract and examine soil mesofauna more precisely. His lifework was carried on by several renowned taxonomists, such as Grandjean, Hammer, Beck, Aoki, Wallwork, Engelbrecht, Corpus-Raros, Lee, Pérez-

Inigo, Baggio, Bhattacharia and Haq (Balogh et al., 2008) with taxonomical descriptions of oribatid mites covering the bigger part of terrestrial habitats. Due to the above researches, nowadays it has become possible to examine oribatid mites from different indication aspects on community level.

One part of the studies on indication possibilities compares natural habitats. In these studies the goal is to reveal spatial and temporal pattern generation characteristics of habitats. Temporal change is examined in few studies (e.g. Irmler, 2006) and in case of spatial examinations different approaches are used: on substrate level (e.g. Fagan et al., 2006; Lindo and Winchster, 2006), examining altitudinal zonation of mountains (e.g. Migliorini and Bernini, 1999; Reynolds et al., 2003; Jing et al., 2005) and only seldom on the level of habitat types (e.g. Balogh et al., 2008). These studies do not always yield consistent results, however, the examinations prove that patterns exist.

Materials and methods

Examination of the suitability of the genus level

Our analysis related to the notion that the genus level does not mean great data loss compared to the species level was carried out based on the databases by Marie Hammer. The work of Hammer was chosen due to the homogeneity and very extensive geographical cover of the databases. The series originate from two different sites of six different countries accordingly (Hammer, 1952, 1958, 1961, 1962, 1966, 1972). Besides the species and genus level the family level was analyzed as well, according to the taxonomical classification in the work by Balogh and Balogh (1992).

Categories of the genus lists

In order to be able to determine to which spatial and temporal distance the oribatid mite genus lists of two samples/sites examined by us correspond, different categories had to be defined. The categories were set up considering which combination of the given spatial and temporal scales the examined genus list pairs originate from. Regarding the time (Ti), we differentiated between 0, 2, 12, 24 and 52 weeks and due to a study (Melamud et al., 2006) we were able to consider six years i.e. 312 weeks, too. In space the smallest distinguishable unit was the different substrate (S), then the different types of habitats/sites (H), the different topographicums (T) follow, and finally the largest unit was the zoogeographic kingdom (K). Substrate is the lowest vegetation level such as soil, förna, leaf litter, moss, bark etc. Site means habitat types such as rainforest, mossy forest, páramo etc. Topographicum is practically a country such as Papua New Guinea or Chile. When differentiating between zoogeographic kingdoms, six kingdoms found in the work by Balogh and Balogh (1992) were considered: Holarktis, Neotropis, Aethiopis, Orientalis, Australis (there Notogea) and Archinotis (there Antarctis), which is the modified version of Müller’s system (1980).

Sources of the genus lists

Genus lists of the different categories were collected from various sources. The first category means the similarity between genus lists of samples collected from the same zoogeographic kingdom (SaK), the same topographicum (SaT), the same type of site (SaH), the same substrate (SaS) and at the same time (Ti-0) (SaK/SaT/SaH/SaS/Ti-0). One part of these genus lists was obtained from our own research. From the soil of a dry oak forest in Törökbálint (Hungary), 9×300 cm^3 förna sample was collected and the mesofauna was extracted from it, the oribatid mites were sorted out and identified to genus level according to the works by Balogh (1965); Balogh and Balogh (1972, 1992); Balogh and Mahunka (1980, 1983) and Olsanowski (1996). Further data for this category were collected by studying the scientific legacy of the late János Balogh, member of the Hungarian Academy of Sciences.

Data for the following four categories were also collected from our research. Samples were collected in 2005 and 2006 in a given quadrat of 100 m^2 in a dry oak forest in Törökbálint, Hungary (N 47°25'38" E 18°54'16") and they were surveyed every two weeks. Every time samples were obtained from three types of substrates: from 500 cm^3 leaf litter, 300 cm^3 förna (from under the leaf litter) and 0.5 dm^2 hypnum moss (*Hypnum cupressiforme*) living on tree trunks. Oribatid mites were extracted with the help of a Berlese-Tullgren funnel (Coleman et al., 2004) and identified on genus level. This examination made it possible to set up categories on pattern levels meaning a distance of two, 12, 24 and 52 weeks, in which substrate (S), site (H), topographicum (T) and zoogeographic kingdom (K) were the same (Sa). Abbreviations of these categories are: SaK/SaT/SaH/SaS/Ti-2, SaK/SaT/SaH/SaS/Ti-12, SaK/SaT/SaH/SaS/Ti-24 and SaK/SaT/SaH/SaS/Ti-52. A study by Melamud et al. (2007) was implied as well, in which samples were collected at different altitudes of Mount Carmel in Israel from the same sites with a difference of six years (312 weeks) (SaK/SaT/SaH/SaS/Ti-312).

Regarding spatial differences, the smallest change in scale is the difference in the substrate: SaK/SaT/SaH/DS/Ti-0, i.e. the substrate is different (D), however, there is no change in time (Ti-0). Genus lists belonging to this category originate from our own database and the above mentioned manuscripts by Balogh. Databases of three further studies were used as well (Behan-Pelletier and Winchester, 1998; Fagan et al., 2006; Lindo and Winchester, 2006).

In case of the following seven categories, only spatial scales "above" substrate change, substrate and time are not differentiated any more so they are marked "X". Abbreviation of the same type of sites which can be found in the same zoogeographic kingdom and in the same topography is SaK/SaT/SaH/XS/Ti-X. Genus lists belonging to this category were obtained from the manuscripts by János Balogh, the study by Migliorini et al. (2005) and the studies by Hammer (1958, 1961, 1962, 1966). Abbreviation of the category of different sites is SaK/SaT/DH/XS/Ti-X. Sources of the series belonging to this category are: studies by Noti et al. (1996); Migliorini et al. (2002); Osler and Murphy (2005); Skubala and Gulvik (2005); Arroyo and Iturrondobeitia (2006); Osler et al. (2006), manuscripts by János Balogh, published series by János Balogh (Balogh et al., 2008) and studies by Hammer (1958, 1961, 1962, 1966). A series belonging here originates from samples collected by Levente Hufnagel in Australia (2006, Australia: QLD, Cairns).

In case of genus lists originating from different topographicums, we considered the point if they originate from the same (SaK/DT/SaH/XS/Ti-X) or different sites (SaK/DT/DH/XS/Ti-X) and if the two topographicums can be found in the same or different zoogeographic kingdoms (DK/DT/SaH/XS/Ti-X, DK/DT/DH/XS/Ti-X). These series come from studies by János Balogh and Marie Hammer.

In the last category only the zoogeographic kingdom can be interpreted as the complete genus lists of the six zoogeographic kingdoms were compared in it according to the work by Balogh and Balogh (1992) (DK/XT/XH/XS/Ti-X).

Data processing methods

The lists created from the Hammer-databases were analysed with Ochiai, Jaccard and Sørensen distance functions and non-metric ordination using the software Syn-tax 2000 (Podani, 2001).

From the other databases we did not consider all possible list combinations which fit the category, only the ones having at least nine genera. After our complete genus list database was set up, the number of genera of the two lists and the number of the common genera were determined considering the genus list pairs in each category. As we had only presence-absence data and the value "d" of the contingency table was not considered in case of the genus list pairs, the Ochiai and Jaccard functions were used as distance functions (Podani, 1997). The similarity data of each category was calculated from the means of the values of the distance functions for the genus list pairs.

As our data were not always independent within a category, it was determined with a complex method to what extent the means of the categories differ from each other. We had several distance function values within each category as we. We had 106 genus list pairs within one category on average. From among the distance function values of each category fifteen values were chosen randomly with the help of a random number generator in the Excel software. It was carried out ten times in case of each category. In this way we got 10 series containing 15 values for each category. Series of the data table containing 10×15 values in case of each of the 14 categories were now independent and since normal distribution could not be observed within each category, the data were analysed with the Kruskal-Wallis statistical test. Each of the 14 series were analysed with the Mann-Whitney post hoc test as well, so we got ten tables containing 14×14 post hoc test results. One table was made out of these ten, which shows 95% confidence interval of the appropriate values of the ten tables. Based on this we were able to decide which categories differ from each other significantly. These statistical tests were carried out using PAST software (Hammer et al., 2001).

The above analyses were carried out on family level as well.

Results and discussion

Controlling the suitability of the genus level

Results of the ordinations carried out for the chosen sites are displayed in *Fig. 1*. For our examination comparisons on genus level are sufficient as switching from species level to genus level did not cause a significant change regarding the distance and position of habitats according to the results of the ordinations. On family level inconsistency is caused by losing information. Using species-level data would be impractical due to taxonomical uncertainty on the one hand and lack of reliable databases on the other hand, and thirdly, due to unjustified increase in distance caused by genera with large number of species.

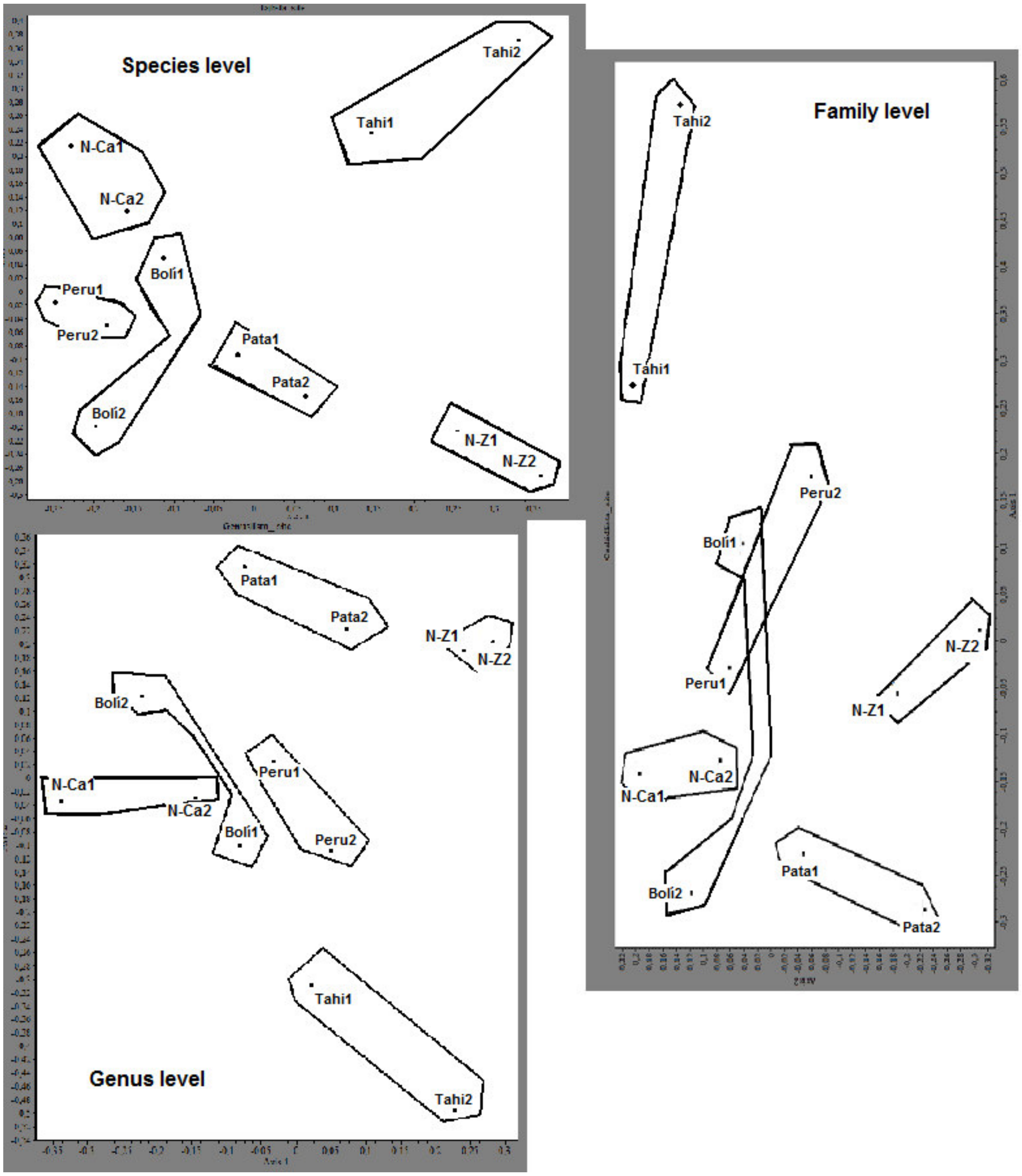

Figure 1. *Non-metric ordination of species, genus and family lists created from Hammer-series. Abbreviations are the following: N-Ca1 and N-Ca2: habitats in North-Canada, Tahi1 and Tahi2: habitats in Tahiti, Pata1 and Pata2: habitats in Patagonia, Boli1 and Boli2: habitats in Bolivia, Peru1 and Peru2: habitats in Peru, N-Z1 and N-Z2: habitats in New-Zealand.*

Order of the genus list categories

As we got nearly the same results using both distance functions (Ochiai and Jaccard), only the results calculated with the help of the Ochiai function are discussed further. *Fig. 2* displays intervals with defined standard error around the Ochiai distance means in case of each category.

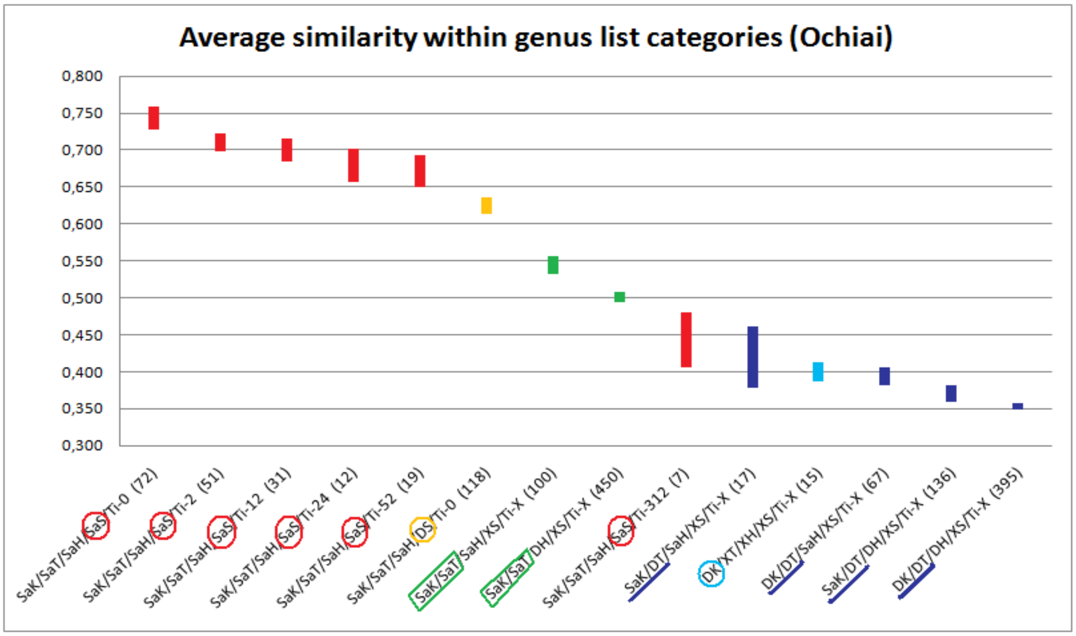

Figure 2. *Average distance within genus list categories using Ochiai function. Next to the category codes, the number of genus list pairs used for calculating the average can be seen in brackets. Remarkable code parts are highlighted*

In *Fig. 2* it can be seen that the largest similarity between samples can be observed in the category where all criteria are the same (SaK/SaT/SaH/SaS/Ti-0) i.e. where the samples were collected at the same time and from the same substrates. Similarities of genus lists originating from different time but the same substrates are the next: first the two-week-difference, then the 12-, 24- and finally the 52-week-difference. Among comparisons on sample level the last one marked with yellow colour is the similarity of genus lists originating from different types of substrates.

As expected, within the same topographicum there is larger similarity between genus lists coming from the same type of sites than between those coming from different sites. This is where the sample series meaning six-year-difference (SaK/SaT/SaH/SaS/Ti-312) turn up. This considerable difference is amazing within one given habitat.

Among the last five there are the four categories in which the difference between lists originating from different topographicums (DT) is measured. Regarding the order of these four categories it is remarkable that the same type of site shows larger similarity than different sites, irrespective of the fact whether the different topographicums are in the same or different zoogeographic kingdoms. This corresponds with the results of a former study conducted with other methods (Balogh et al., 2008). The category DK/XT/XH/XS/Ti-X fits in the above mentioned categories in the order. Using the Jaccard distance function this is the only category position that changes places with the category DK/DT/SaH/XS/Ti-X.

The order set up with the help of genus lists based on the complete database met our expectations, so it can be definitely an appropriate reference in indication researches.

Table 1. The significance of differences between genus list categories according to Mann-Whitney tests. (Due to volume reasons the 14x14 table was divided vertically into two and put under one another.)

Ochiai	SaK/SaT/SaH/SaS/ Ti-0	SaK/SaT/SaH/SaS/ Ti-2	SaK/SaT/SaH/SaS/ Ti-12	SaK/SaT/SaH/SaS/ Ti-24	SaK/SaT/SaH/SaS/ Ti-52	SaK/SaT/SaH/DS/ Ti-0	SaK/SaT/SaH/XS/ Ti-0
SaK/SaT/SaH/SaS/ Ti-0	×	0,127-0,365	0,088-0,277	0,032-0,325	0,000-0,319	0,000-0,199	0,000-0,064
SaK/SaT/SaH/SaS/ Ti-2	0,732-1,000	×	0,275-0,620	0,086-0,454	0,051-0,418	0,016-0,067	0,000-0,009
SaK/SaT/SaH/SaS/ Ti-12	0,695-1,000	1	×	0,220-0,625	0,175-0,530	0,056-0,289	0,000-0,017
SaK/SaT/SaH/SaS/ Ti-24	0,621-1,000	0,893-1,000	1	×	0,204-0,571	0,050-0,418	0,003-0,030
SaK/SaT/SaH/SaS/ Ti-52	0,619-1,000	0,918-1,000	1	1	×	0,163-0,531	0,012-0,109
SaK/SaT/SaH/DS/ Ti-0	0,399-0,938	0,700-1,000	0,798-1,000	0,798-1,000	1	×	0,072-0,482
SaK/SaT/SaH/XS/ Ti-0	0,017-0,442	0,069-0,591	0,122-0,673	0,308-0,818	0,368-0,940	0,656-1,000	×
SaK/SaT/DH/XS/ Ti-0	0,000-0,122	0,000-0,113	0,000-0,149	0,045-0,585	0,051-0,555	0,495-0,982	0,803-1,000
SaK/SaT/SaH/SaS/ Ti-312	0,000-0,037	0,000-0,001	0,000-0,002	0,082-0,277	0,000-0,048	0,019-0,267	0,639-1,000
SaK/DT/SaH/XS/ Ti-X	0,000-0,021	0,001-0,003	0,001-0,005	0,003-0,018	0,009-0,061	0,109-0,551	0,780-1,000
DK/XT/XH/XS/Ti-X	0,05>>	0,05>>	0,05>>	0,05>>	0,000-0,007	0,000-0,001	0,013-0,519
DK/DT/SaH/XS/Ti-X	0,000-0,002	0,000-0,001	0,000-0,001	0,000-0,001	0,001-0,008	0,000-0,019	0,148-0,745
SaK/DT/DH/XS/Ti-X	0,00-0,005	0,000-0,001	0,000-0,001	0,000-0,001	0,000-0,006	0,000-0,016	0,248-0,845
DK/DT/DH/XS/Ti-X	0,000-0,001	0,000-0,001	0,000-0,002	0,000-0,002	0,000-0,002	0,000-0,005	0,000-0,137

Table 1. cont.

Ochiai	SaK/SaT/DH/XS/ Ti-0	SaK/SaT/SaH/SaS/ Ti-312	SaK/DT/SaH/XS/ Ti-X	DK/XT/XH/XS/Ti-X	DK/DT/SaH/XS/Ti-X	SaK/DT/DH/XS/Ti-X	DK/DT/DH/XS/Ti-X
SaK/SaT/SaH/SaS/ Ti-0	0,000-0,001	0,05>>	0,05>>	0,05>>	0,05>>	0,05>>	0,05>>
SaK/SaT/SaH/SaS/ Ti-2	0,000-0,001	0,05>>	0,05>>	0,05>>	0,05>>	0,05>>	0,05>>
SaK/SaT/SaH/SaS/ Ti-12	0,000-0,002	0,05>>	0,05>>	0,05>>	0,05>>	0,05>>	0,05>>
SaK/SaT/SaH/SaS/ Ti-24	0,000-0,018	0,000-0,003	0,05>>	0,05>>	0,05>>	0,05>>	0,05>>
SaK/SaT/SaH/SaS/ Ti-52	0,000-0,008	0,000-0,001	0,000-0,001	0,05>>	0,05>>	0,05>>	0,05>>
SaK/SaT/SaH/DS/ Ti-0	0,016-0,151	0,000-0,003	0,001-0,006	0,05>>	0,05>>	0,05>>	0,05>>
SaK/SaT/SaH/XS/ Ti-0	0,255-0,673	0,000-0,398	0,065-0,334	0,000-0,022	0,000-0,047	0,004-0,017	0,000-0,002
SaK/SaT/DH/XS/ Ti-0	×	0,116-0,462	0,240-0,694	0,000-0,258	0,000-0,331	0,000-0,295	0,002-0,037
SaK/SaT/SaH/SaS/ Ti-312	1	×	0,382-0,732	0,025-0,348	0,048-0,503	0,105-0,533	0,000-0,139
SaK/DT/SaH/XS/ Ti-X	1	1	×	0,061-0,331	0,177-0,533	0,174-0,467	0,002-0,164
DK/XT/XH/XS/Ti-X	0,667-1,000	0,418-0,939	0,722-1,000	×	0,261-0,599	0,271-0,577	0,092-0,330
DK/DT/SaH/XS/Ti-X	0,706-1,000	0,514-1,000	1	1	×	0,270-0,508	0,053-0,386
SaK/DT/DH/XS/Ti-X	0,659-1,000	0,741-1,000	0,785-1,000	1	1	×	0,042-0,116
DK/DT/DH/XS/Ti-X	0,182-0,754	0,321-0,893	0,715-1,000	0,581-1,000	0,704-1,000	1	×

Significance of distances between genus list categories

In *Table 1* the 95% confidence intervals of the Mann-Whitney post hoc tests are shown. Cells marked with grey colour show significant difference between the two given categories. The heavy black frame means deviation from the values of the tables set up using Jaccard functions, regarding the existence of significance.

Our first remarkable result is that the average distances between genus lists originating from the same substrate but from different dates within a year does not differ from each other significantly. According to our former assumptions time difference could have been detected regarding a one-year-difference, however, it could not be detected from the substrates of the temperate dry oak forest studied. Consequently, if genus lists of the complete habitat type were examined with time lags less than a year, no change could be detected on site level, either. Irmler (2006) had a similar result on species level in a long-term European study.

The difference between oribatid mite communities originating from different substrates (DS/Ti-0) can be significantly larger in case of certain distance functions than the difference between communities coming from the same type of substrate (SaS/Ti-0). Besides, there is no large difference between samples collected within a year from a given type of substrate. It follows from these two statements that if genus lists originate from different types of substrates, there is larger difference between them than if samples are collected within a year from the same type of substrate. Consequently, the quality of substrate in a given habitat type is a more important factor in the composition of the oribatid mite community than time changes within a year.

Similarity between oribatid mite genus lists of the same types of sites (SaK/SaT/SaH/XS/Ti-0) differs significantly from the similarity between genus lists originating from the same type of substrate (SaK/SaT/SaH/SaS) if samples were collected with a time lag of maximum 24 weeks in the latter case. At the same time, the distance between genus lists coming from different types of substrates is similar to the distance between oribatid mite genus lists of the same or different habitats in a given topographicum, i.e. the type of substrates plays a similarly important role in the quality of the oribatid mite community as habitat types in a given topographicum.

The distance between genus lists originating from the same or different types of substrate is much smaller than the distance between genus lists of different topographicums (XK/DT/...), independently of the fact if sites in the same or different zoogeographic kingdoms are compared. More remarkable is the fact that the similarity of genus lists coming from the same sites in the same topographicums does not differ significantly from the similarity of genus lists originating from the same sites in different topographicums. It means that oribatid mite communities of the same types of habitats resemble each other nearly in the same way no matter if they originate from the same or different topographicums.

Order of the family list categories and similarity between zoogeographic kingdoms

Analysis on family level does not differ much from that on genus level. Results are displayed in *Fig. 3*. Results of the analyses carried out with both distance functions coincided completely here, therefore only results calculated with the help of Ochiai function are discussed. The most remarkable difference is the increase in similarity between lists of the zoogeographic kingdoms, i.e. the category DK/XT/XH/XS/Ti-X came before the category in which there is a difference only in substrates

(SaK/SaT/SaH/DS/Ti-0). Another change is the fact that samples with a difference of one year (52 weeks) moved forward to appear between the categories with a time difference of two and 12 weeks. Besides, the category meaning a time difference of six years (312 weeks) moved one place forward in comparison with its place in the order of genus lists. Similarities between the family and genus level point out that it is possible to use the family level instead of the genus level from a taxonomical point of view in the comparisons suggested by us. This corresponds to the results of the meta-analysis by Osler and Beattie (1999) similar to ours, in which it was found after analysing 25 studies that habitat is mainly chosen on family level and they suggested that the family level can be enough to quickly estimate the diversity of an area.

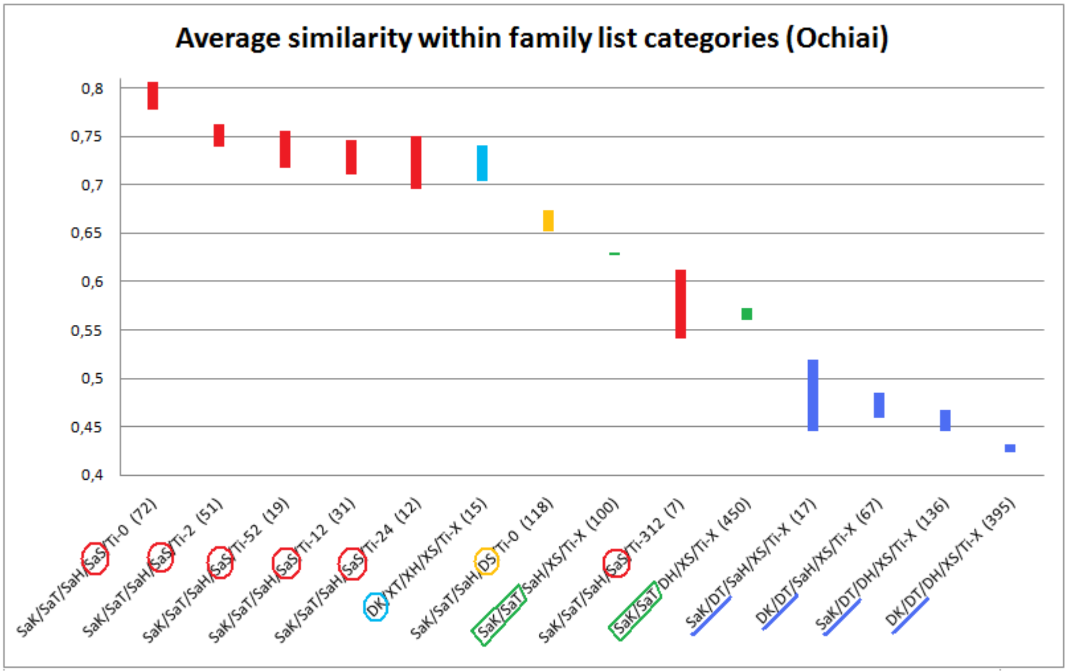

Figure 3. *Average distance within family list categories using Ochiai function. Next to the category codes, the number of family list pairs used for calculating the average can be seen in brackets. Remarkable code parts are highlighted.*

Complete genus lists of zoogeographic kingdoms resemble each other so much as if site lists originating at least from different countries (topographicums) were compared (*Fig. 2*). It is remarkable from the point of view that zoogeographic kingdoms were mainly differentiated based on vertebrate groups and if an invertebrate group, in this case oribatid mites are regarded, difference between zoogeographic kingdoms on vertebrate level can cover smaller topographicums and not continents in case of oribatid mites. On family level, distance between zoogeographic kingdoms means the same difference as it is between family lists originating from the same or different types of substrate. It means that zoogeographic kingdoms do not differ from each other to a great extent on family level. In case of oribatid mites, zoogeographic kingdom is not a reasonable unit of differentiation on family level, while it is one of the units of differentiation in case of vertebrates, which is most likely due to the fact that separation of oribatid mite families preceded historically the separation of continents.

Conclusions

Category orders as results of the analyses mostly met our expectations, first of all in case of genus lists, i.e. larger distances between genus lists correspond to larger spatial scales. However, time difference within a year could not be pointed out to a considerable extent either on genus or on family level. The most important differences in the orders are the followings:

1. the difference between samples originating from the same and different types of substrate;
2. the difference between the distance of lists coming from the same type of substrate in a given site and the distance of genus lists originating from the same topographicum but from different or the same types of site;
3. the difference between the distance of lists coming from the same or different types of substrate in a given site and the distance of lists originating from different topographicums;
4. the difference between the distance of lists coming from the same types of sites in a given topographicum and the distance of lists originating from different topographicums.

The analysis on family level differs from that on genus level that much that family lists of the zoogeographic kingdoms resemble each other as much as family lists originating from the same types of substrate.

Thus we compiled a reference list with the help of which it can be expressed to what spatial distances the similarity – shown with distance functions – of the genus or family lists of two oribatid mite communities originating from samples of unknown quality corresponds. Disturbed and transformed habitats can be compared with the help of oribatid mite communities based on an existing reference list.

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EFFECTS OF MUNICIPAL SEWAGE ON THE GROWTH PERFORMANCE OF *CASUARINA EQUISETIFOLIA* (FORST. & FORST.) ON SANDY SOIL OF EAST COAST AT KALPAKKAM (TAMIL NADU, INDIA)

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Abstract. Municipal sewage both untreated and treated, that pollutes the water bodies was used growing Casuarinas (*Casuarina equisetifolia* Forst. & Forst.), an exotic bio-shield tree species on sandy soil at Kalpakkam on the East Coast of Indian peninsula; it may help in reducing aquatic pollution as well as in preventing the high tides like Tsunami. Significant increase in growth performance in plant-height, branches, root length and the biomass was recorded in the saplings irrigated with untreated municipal raw sewage (RS) and treated sewage (TS) compared to that irrigated with unpolluted potable water (PW) over a period of 13 months, October 2005 to October 2006. These growth parameters showed close relationship with the nutrient contents of municipal RS, TS and PW, the former being characterized by relatively higher pH, electrical conductivity (µs/cm), total dissolved solids, total suspended solid, total hardness, chloride, sulphate, biochemical oxygen demand, chemical oxygen demand, calcium, magnesium, sodium, potassium, bicarbonates, total alkanity, nitrate, phosphate and carbonates (mg/l) compared to that of the TS and PW indicating profound influence of municipal sewage on the plant growth.

Key words: *Casuarinas; growth; municipal raw sewage; treated sewage*

Introduction

Land application of municipal sewage is a cost-effective method of its treatment and disposal. Wastewater irrigation and the land application of sewage treatment residues (sludge or bio-solids) are traditional practices around the world. In Paris sewage farms existed as early as 1868. The use of sewage effluents for the irrigation of government farms in Egypt has been in implementation since 1915. A survey of current wastewater use practices in developing countries carried out by the United Nations Development Program and World Bank (1990) estimated that some 80% of the wastewater from urban areas in developing countries is currently used for permanent or seasonal irrigation (Cooper, 1991). Untreated wastewater was used to irrigate at least five lakh hectares in Latin America, with over half of this area in Mexico (Rodriguez et al.,1994). Land application using untreated and treated municipal sewage for biomass-production has many benefits (Idelovitch and Michael, 1984; Juwarkar et al.,1995), besides preventing pollution of lotic and lentic ecosystems. Sophisticated treatment technologies like trickling filters, activated sludge processes, aerated lagoons, bio-methanation and incineration, etc. are energy consuming and cost-intensive and beyond the reach of many developing countries. Further, the discharge of treated wastewater enriched with nutrients and other pollutants can cause eutrophication in both lotic and lentic systems. This warrants adoption of an appropriate wastewater management system where in twin benefits of treatment and recycling and reuse can be achieved, and ecological development can be promoted.

China and India make significant use of wastewater (Bartone, 1991). Raw sewage used for irrigation in India over a 15-year period was reported to have improved the soil structure (Das and Kaul, 1992; Mathan, 1994), and increased soil nutrients and organic carbon content without increasing heavy metals to toxic levels (Gupta et al.,1998). About 6350 million cubic meter of wastewater is being generated every year from 212 class I and 242 class II towns in India, of which only 36% in class I cities and 14% in class II towns are collected due to limited treatment facilities (Thawale et al., 2006). Rapid infiltration of sewage into sandy soils can result in improvement of its physical, chemical and microbiology quality. The soil and its associated ecosystem components act as physico-chemical and bio-reactors capable of treating or stabilizing pollutants of liquid and solid origin through degradation, adsorption and utilization by trees. The nutrients like nitrogen, phosphates and potassium of sewage along with the micronutrients as well as organic matter present in municipal sewage could be added advantageously to add to the fertility of the soil, along with the irrigation potential of the water. However, application of untreated effluents to land can be carried out with certain precautions as it is not completely free from environmental health risk.

The potential for adverse health impacts of irrigation with wastewater has been addressed in a number of earlier studies. The passage through field vegetation and/or the filtration that accompanies irrigation and subsequent runoff and drainage is expected to reduce the level of parasites and other harmful microorganisms, in addition to the observed changes in chemical concentrations. Effective and appropriate wastewater treatment processes can reduce the health hazards associated with wastewater use. However, Feachem et al. (1983) showed that treated effluent coming through stabilization ponds or conventional treatment plants followed by maturation ponds or sand filtration may be free, or almost free of pathogens.

Untreated sewage was used to irrigate forestry plantations *Tamarix aphylla*, *Eucalyptus camaldulensis* and *Acacia salicina* in Kuwait (Armitage, 1985). Braatz, Kandiah reported that farms such as the El-Gabal El-Asfar located near Cairo where tree plantations of 200 ha irrigated with wastewater was established since 1911 to dispose of the city's untreated sewage (Braatz, 1996). Pioneering studies on the application of treated municipal wastewater on forest lands as a means of purification and groundwater recharge were carried out in central Pennsylvania (U. S. A.) during 1963 to 1977 (F.A.O., 1978). In Murray Darling Basin (New South Wales, Australia) forest tree plantation irrigated with secondary treated wastewater has increased from 500 ha in 1991 to about 1500 ha in 1995 during 1991 to 1997; there are now more than 60 effluent-irrigated tree plantations varying in size from one to several hundred hectares (CSIRO,1995). Braatz and Kandiah (1996) reported that the secondary treatment sewage effluent was applied on three different forest areas of mixed hardwood stand consisting mainly of oaks (*Quercus spp.*), red pine (*Pinus resinosa*) and white spruce (*Picea glauca*). Tree plantations can remove nutrients such as nitrogen and phosphates from land-applied wastewater for their growth without any harmful effect on the environment (Stewart et al., 1990). Using municipal effluent to grow trees in suburban areas in developing countries may help solve the dual problems of wastewater disposal and lack of fuel wood supply (Paliwal et al., 1998; Singh and Bhati, 2004). Banerjee et al. reported on the effects of distillery effluents in the growth of C.

equisetifolia (Banerjee and Bahal, 2004). However, large scale use of municipal sewage/wastewater for the irrigation of tree-plantations is still relatively limited. Casuarinas *(C. equisetifolia)* or Babool *(A. nilotica*) are important species for fuel wood, and the former being planted now all along the east coast of Indian peninsula as bio-shield to prevent the high tides like Tsunami. The present study reports the growth performance of C. *equisetifolia* irrigated with municipal RS and TS in relation to that of near by PW on the nutrient poor sandy soil at Kalpakkam.

Materials and methods

Description of the study area

The present study was carried on the sandy soil at Kalpakkam (12º 30' N and 80º 10' E) on the east coast of India in the Kancheepuram district of Tamil Nadu. Experimental plots each of size 8 × 5 sq ft were prepared with randomized block design with each treatment comprising of six replicates and separate irrigated channels of municipal RS, TS and PW,. Six saplings of casuarinas were planted in each of the replicate plots. Municipal RS and TS from the Extended Aeration Activated Sludge System of Kalpakkam and PW were channeled to these plots for irrigation for a period of 13 months starting from October 2005 to October 2006.

Physico-chemical analysis

The physico-chemical characteristics – pH, electrical conductivity (µs/cm), total alkalinity, total hardness, total dissolved solids, total suspended solids, 5-days biochemical oxygen demand, chemical oxygen demand, nitrate nitrogen, chloride, sulfate, phosphate, sodium, potassium, calcium, magnesium, bicarbonates and carbonates (mg/l) of municipal RS, TS and that of PW were analyzed using methodology described in (APHA, 1998).

Plant growth measurement

The heights and number of branches of each casuarinas sapling in each replicate plot were recorded every month, and the root length and biomass of each of the saplings were measured at the end of 13 months of the experiment.

Statistical analysis

ANOVA analysis of the data on height of the saplings grown in municipal RS, TS and PW over 13 months period was computed. Multiple correlation analysis was computed between the height of the saplings grown in RS, TS and PW and the nutrients quality and quantity of sewage and PW over the 13 months period.

Results

Water-quality parameters

Analyses of water quality parameters such as pH, electrical conductivity, and total alkalinity, hardness, dissolved solids (TDS), suspended solids (TSS), Biochemical Oxygen Demand$_5$, chemical oxygen demand, nitrate nitrogen, chloride, sulfate, phosphate, sodium, potassium, calcium, magnesium, bicarbonates and carbonates (mg/l) of municipal RS, TS and PW used for irrigation of casuarinas saplings showed that the average concentration of each of the above water quality parameters of municipal RS were higher than that of the TS, both of which were higher than that of the PW (*Table 1*). The average temperature in the RS, TS and PW were 28.1, 28.0 and 27.0 ºC, respectively. The average pH and that of electrical conductivity, TDS, total hardness, chlorides, sulphate, calcium, magnesium, sodium, total alkanity, bicarbonates and nitrates (mg/l) in RS were higher than that of TS and PW. The average TSS, COD, BOD, phosphates, potassium and carbonates in RS were more than six, two, eleven, two, two and four folds higher than that of the TS, respectively and were ninety one, three, sixteen, eight, three and four folds higher than that of the PW, respectively (*Table 1*).

Table 1. *Mean concentration of raw sewage, treated sewage and that of potable water which was used for irrigation*

Physico-chemical parameters	**Raw sewage**	**Treated sewage**	**Potable water**
Temperature (ºC)	28.1	28	27
pH	7.1	6.8	6.73
Electrical Conductivity (µs/cm)	924.6	731.8	613
TDS (mg/l)	584.5	422.8	368.5
TSS (mg/l)	377.1	61.8	4
Total hardness (mg/l)	165.8	109.8	98
Chloride (mg/l)	207.12	131.6	118
Sulphate (mg/l)	33.5	31.3	22
COD (mg/l)	450.2	157.9	133.4
BOD (mg/l)	244.5	20.2	16.5
Ca (mg/l)	37	26.8	20.4
Mg (mg/l)	20.5	11.5	10.7
Sodium (mg/l)	129.4	79.7	41.2
Potassium (mg/)	37	17.1	10.4
Bicarbonates (mg/l)	147.9	95.1	82.1
Total alkanity (mg/l)	198.1	115.8	101.4
Nitrates (mg/l)	34.5	19.5	14.5
Phosphates (mg/l)	1.62	0.7	0.2
Carbonates (mg/l)	2.82	0.7	0.62

Casurina equisetifolia sapling growth

One of the ways to reduce the pollution of the receiving water bodies due to the municipal sewage is its optimum reuse in irrigation of tree plantations. Bio-shield exotic tree, *C. equisetifolia* is widely grown now on the east coast of Indian peninsula to prevent high tides like tsunami waves that lashed the coast during December 26, 2004, and the use of the municipal untreated and treated sewage for irrigation and watering of these tree nurseries and plantations may be useful in better establishment and quick growth of the plantations. Our study revealed that the height of *C. equisetifolia* prior to the irrigation with municipal RS, TS and PW were 25.95 ± 2.5, 28.4 ± 4.9 and 25.3 ± 5.8 cm, respectively. After a month of irrigation with RS, TS and PW the height of the saplings increased to 29.6 ± 3.2, 32.2 ± 5.6 and 27.5 ± 7.0 cm respectively, whereas after three months of irrigation, that of the saplings increased to 52.4 ± 8.1, 44.9 ± 9.8 and 41.0 ± 13.6 cm respectively, after six months of irrigation that has increased further to five, four and four folds, respectively, and after nine months of irrigation that increased to eight, six and six folds, respectively while after twelve months of irrigation that increased to eleven, eight and eight folds, respectively *(Figure 1)*. ANOVA analysis of the plant height grown across the treatments of RS, TS and PW showed significant difference in the height of the saplings between and within treatments, being recorded highest in the former and minimum in the later (*Table 2*). There was significant positive correlation between the height of the saplings and the quality of municipal sewage i.e., RS and TS in the present study ($R^2 = 0.74$, $p < 0.01$). The lower height in the saplings irrigated with PW were most probably due to its relatively low nutrient contents.

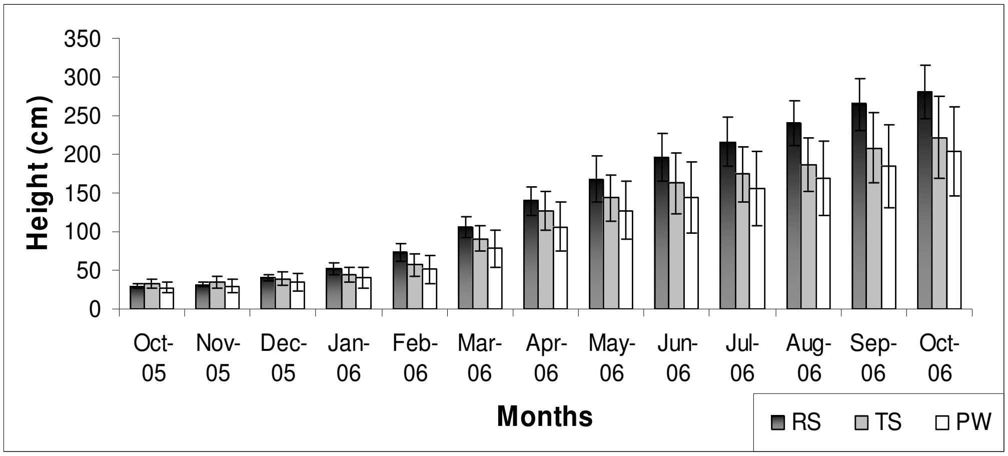

Figure 1. *Variation in the height of the C. equisetifolia across the three different treatments*

Number of individual branches of the *C. equisetifolia* did not appear till the third month of irrigation with municipal RS, TS and PW. After six months of irrigation, the number of branches per sapling was 1.5 + 1.1, 1.5 + 0.9 and 0.1 + 0.3, respectively, and after nine months their cumulative numbers per sapling increased to 2.3 + 0.9, 1.8 + 0.8 and 1.1 + 1.0, respectively, while after twelve months that was further increased to 2.3 + 0.9, 1.5 + 0.5 and 1.3 + 0.4, respectively *(Figure 2)*.

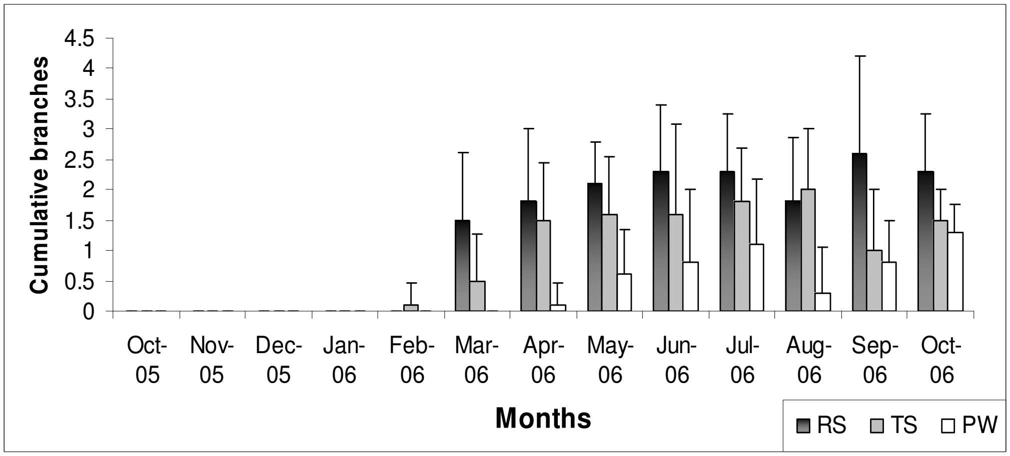

Figure 2. *Variation in the number of branches of the C. equisetifolia across three different treatments*

Table 2. *ANOVA analysis of height of the plants (C. aquesetifolia) across different irrigation treatments*

Source of Variation	**df**	**SS**	**MS**	**F**	**P-value**	**F crit**
Height of the plants:						
Between treatments	12	205859.6	17154.97	75.07326**	5.11E-16	2.183377
Within treatment	2	9264.416	4632.208	20.27139**	6.99E-06	3.402832
Error	24	5484.232	228.5097			
Total	38	220608.2				

**P<0.01

The root length of the saplings of *C. equisetifolia* prior to the treatment with municipal RS, TS, and PW was 14 ± 1.8, 13.1 ± 1.5 and 13.9 ± 3.3 cm respectively, which after 13 months of irrigation, has increased to more than four folds in RS, to about three folds in TS and about three folds in PW *(Figure 3)*.

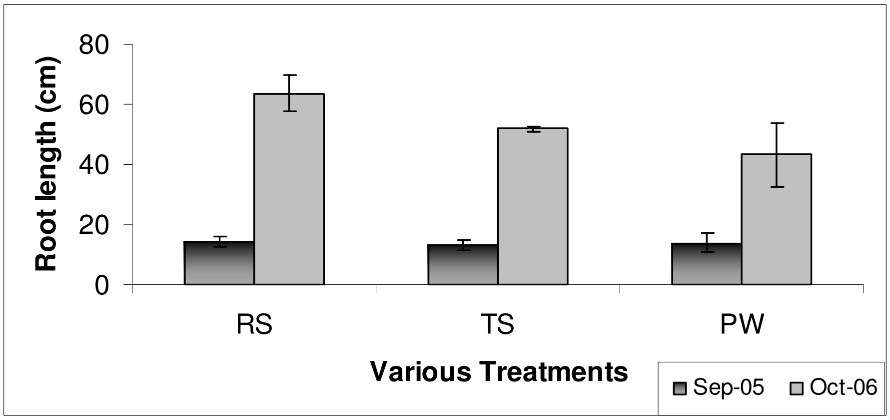

Figure 3. *Variation in root length across different treatments*

Similarly, the average wet biomass of each sapling after 13 months of irrigation of municipal RS, TS and PW were 3.5, 2.4 and 2.0 kg per sapling, respectively *(Figure. 4)*.

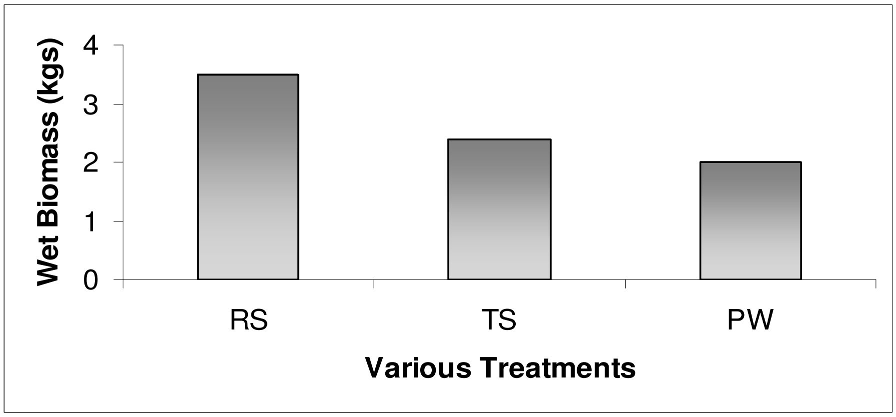

Figure 4. *Variation of the wet biomass (kg) of the C. equisetifolia across different treatments*

Discussion

The slight increase in temperature in RS recorded during the present study was probably because of its relatively higher microbial activity. Besides, in consistence to the present findings, Singh and Bhati (2003) reported higher concentration of various nutrients in municipal sewage compared to that of the canal water. The decrease in average concentration of the physico chemical parameters of the TS compared to that of RS in the present study was because of extended aeration activated sludge treatment (Balluz et al., 1977). In consistence to the present findings, (Fatma et al., 1998) reported a decrease in physico-chemical concentration in treated wastewater used for irrigation in Egypt. Similarly, (Alhumoud et al., 2003) reported decrease in average values of the water quality parameters – TSS, TDS, BOD, COD, pH, NH_3, phosphates, temperature (°C) in secondary and tertiary wastewater in Kuwait.

The significant increase in the sapling height in the treatment irrigated with municipal RS was probably due to availability of increased organic matter, and both macro- and micro- nutrients, especially total and available N in the municipal sewage (Braatz and Kandiah, 1996). Omran et al. (1998) in corroboration to the present findings, also reported increased growth density and shoot length in Navel Orange trees irrigated with sewage in Egypt. Similarly, increase in water and nutrient availability through effluent application influenced the growth of *A. nilotica* (Singh and Bhati, 2004; Singh and Singh, 2000), that of Eucalyptus (Singh and Bhati, 2003), *Hardwickia binata* (Paliwal et al., 1998), *E. grandis* (Stewart et al., 1990) and *P. radiate* (Sheriff et al., 1986). The sewage or drainage water used after primary treatment for irrigation increased the growth of woodlots with most commonly used species, *C. glauca, E. camaldulensis* and *T. aphylla* (El-Lakany, 1995). In India utilizing raw sewage for irrigation of plantations of *E. tereticornis, Populus deltoides* and *Leucaena leucocephala* showed higher growth than that of the trees irrigated with well water; the eucalyptus being six percent taller after 48 months; the leucaena being 12 percent taller after 36 months; and the poplar being four percent taller after 24 months (Das and Kaul, 1992).

The increase in the number of branches of saplings grown in municipal sewage was probably because of availability of relatively more nutrients present in the sewage compared to that of PW. The availability of water and nutrients probably had positive effects on root growth (Singh and Bhati, 2003; 2004). This may be due to the response of casuarinas saplings to the nutritive elements present in the sewage water. Previous studies showed that the increase in plant growth and yield was primarily related to increased phosphorus uptake (Silber et al., 2005). Thus, increase in the growth performance and biomass of *C. equisetifolia* irrigated with municipal RS and TS compared to that of the PW during the present study is most probably due to the constant supply and continuous replenishment of nutrients like nitrogen and phosphorous to the saplings and improved soil structure in the rhizosphere. (Breaux et al., 1995) in corroboration, reported that addition of sewage with influx of nutrients stimulated biomass production being beneficial to soil formation; Singh and Bhati (2003; 2004) reported higher growth and biomass in seedling of acacia and eucalyptus respectively, which they attributed to the effects of available nutrients, particularly N in the effluent facilitating leaf initiation that converted more solar energy enhancing CO_2 fixation and photosynthate level leading to higher growth and biomass production (Braatz and Kandiah, 1996).

The higher N conditions result in higher growth of leafy shoots through reinvestment of assimilates towards photosynthetic parts (Li et al., 1991).

Conclusion

Municipal sewage was rich in nutrients; its use in growing tree-plantation is an alternative treatment aimed at achieving the stringent standards for wastewater disposal into the receiving water bodies. Raw sewage and treated sewage can be used for irrigation for growing the bio-shield tree, *C. equisetifolia* on the poor sandy soil of east coast of Indian peninsula as the raw and treated sewage showed increased the growth of the tree during the present study.

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