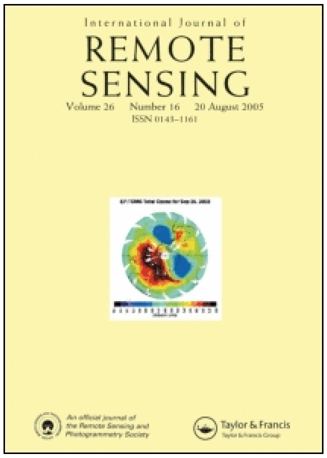

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Accurate prediction of bird species richness patterns in an urban environment using Landsat-derived NDVI and spectral unmixing

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Accurate prediction of bird species richness patterns in an urban environment using Landsat-derived NDVI and spectral unmixing

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Urban landscapes are expanding rapidly and are reshaping the distribution of many animal and plant species. With these changes, the need to understand and to include urban biodiversity patterns in research and management programmes is becoming vital. Recent studies have shown that remote sensing tools can be useful in studies examining biodiversity patterns in natural landscapes. The present study aimed to explore whether remote sensing tools can be applied in biodiversity research in an urban landscape. More specifically, the study examined whether the Landsat-derived Normalized Difference Vegetation Index (NDVI) and linear spectral unmixing of urban land cover can predict bird richness in the city of Jerusalem. Bird richness was sampled in 40 1-ha sites over a range of urban environments in 329 surveys. NDVI and the per cent cover of built-up area were strongly and negatively correlated with each other, and were both very successful in explaining the number of bird species in the study sites. Mean NDVI in each site was positively correlated with the site bird species richness. A hump-shaped relationship between NDVI and species richness was observed (when calculated over increasing spatial scales), with a maximum value (Pearson's R=0.87, p<0.001, n=40) at a scale of 15 ha. We suggest that remote sensing approaches may provide planners and conservation biologists with an efficient and cost-effective method to study and estimate biodiversity across urban environments that range between densely built-up areas, residential neighbourhoods, urban parks and the peri-urban environment.

1. Introduction

The rapid expansion of urban landscapes and their human populations are generating increasing interest in the patterns and processes shaping biodiversity in these human-dominated landscapes (Marzluff and Ewing 2001, Faeth *et al*. 2005, Shochat *et al*. 2006). Urbanization, together with the reshuffling of species by humans, is often related to a process termed biotic homogenization (Jokimaki *et al*. 1996, Blair 2001, Lockwood and McKinney 2002, Olden 2006), where few species

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are becoming widespread in many parts of the globe. However, cities are not uniform, and are composed of different regions with different types and strengths of human impact. Many cities show a decline in urbanization levels along a gradient ranging from their core to their edges (Pickett *et al.* 2001). Several studies have examined biodiversity changes across urban gradients (McDonnell and Pickett 1990, Alberti *et al.* 2001, Marzluff *et al.* 2001, Chace and Walsh 2006). In general, it has been shown that even very mobile animals often show substantial spatial variation in their distribution within the city (Marzluff 2001). A common general pattern seen is one of declining diversity from the city periphery to the core (McDonnell and Pickett 1990, Clergeau *et al.* 1998, Cam *et al.* 2000, Melles *et al.* 2003)

As reviewed by Marzluff *et al.* (2001), urban studies focusing on birds have increased steadily since Pitelka (1942) examined the effects of human development on bird abundances. Birds provide an especially valuable group for examining the effect of urban gradients for several reasons: there are many bird species in the city, they can be identified with relative ease, are widespread and show high spatial variation. As such, birds provide an opportunity for an assessment of the urban effects on biodiversity patterns along environmental and spatial gradients (Savard and Falls 1981, Drasch *et al.* 1987, Blair 1999, 2004).

The increasing availability of remotely sensed observations facilitates the development of new tools and approaches for understanding the urban environment (Ben-Dor *et al.* 2001, Miller and Small 2003). Satellite-derived vegetation indices are excellent estimates of productivity and can also quantify spatial heterogeneity of vegetation, two important factors shaping biodiversity patterns (Tucker and Sellers 1986, Mittelbach *et al.* 2001). Several recent studies have demonstrated the potential of using remote sensing approaches in predicting species richness in natural landscapes (e.g. Bawa *et al.* 2002, Luoto *et al.* 2002, Oindo and Skidmore 2002, Willis and Whittaker 2002, Honnay *et al.* 2003, Seto *et al.* 2004, Nichol and Lee 2005, St-Louis *et al.* 2006, Levin *et al.* 2007). Nevertheless, a recent literature review that included over 120 studies in which satellite images were used for avian applications (Gottschalk *et al.* 2005) showed that few studies used satellite images when focusing on avian biodiversity in urban environments (Berry *et al.* 1998, Alberti *et al.* 2000, Mörtberg and Wallentinus 2000). This may be due to the fact that high spatial resolution satellite images have become widely available only since 1999 with the launch of IKONOS, and that prior to NASA's Geocover dataset (Tucker *et al.* 2004) there were very few, if any, freely available Landsat images globally. In fact, we found very few published studies (e.g. Alberti *et al.* 2000) that examined the relationship between remotely sensed derived land cover and bird species richness. Although urban areas are usually accessible, their increasing size and dominance in human societies, in addition to the high cost and time required for detailed field surveys over large areas, make remote sensing a potentially useful tool for improving our ability to monitor urban areas.

In this work, we aimed to examine the feasibility of Landsat-derived remote sensing analyses (Normalized Difference Vegetation Index (NDVI) mapping and linear spectral unmixing) for predicting bird species richness in an urban landscape. We compare changes of bird richness along two gradients within an urban landscape: (1) a geographical periphery–core gradient from the edge of the city inwards and (2) an urbanization gradient from the most built up and urbanized regions (the downtown business district) to urban parks within the city and the city's outskirts.

2. Methods

2.1 *Study area*

The study was conducted in Jerusalem, which is the largest city in Israel in terms of its area size (113.4 km^2) and its number of residents (729 900 in 2006, compared with only 21 000 people in 1875 and 260 000 people in 1944 (Amiran and Shahar 1961, The Central Bureau of Statistics 2006)). The city is located at the transition between the desert and Mediterranean climates. Due to its hilly topography, several valleys reach from the city outskirts inwards (Efrat and Noble 1988), forming green corridors within the city. The building regulation legacy of the British mandate period (1917–1948) currently includes limits on building height and a ban on industrial construction within the city (Efrat and Noble 1988). Some areas, mainly in the city's valleys, were defined as green belts in the city plans of 1930 and 1944, with gardens and greenery being established in various neighbourhoods (Kendall 1948, Kark 1991, Cohen 1994). As of today, some of these valleys have remained as open spaces surrounded by residential neighbourhoods. The green corridor along which we chose to examine changes in diversity is the largest in the city and runs from the south-western city periphery to the inner downtown region (figure 1). Today, it is more continuous than several other corridors, although interrupted by a few roads, a narrow residential area and a commercial area, as marked in figure 1. This corridor was defined as a green corridor already in the zoning plans of 1930 and 1944 (Kendall 1948). Additional open spaces around this corridor have been added in subsequent city plans (e.g. the 1959 city plan).

2.2 *Sampling design*

Overall, 40 sampling sites were selected in the city (figure 2, table 1). Four main types of urban environments were defined. These included the (1) downtown area, a commercial core area that is mostly built up and paved and has very little vegetation; (2) residential areas, sampled in all cases in a secondary street of a similar width at a constant distance from the nearest intersection of roads; (3) urban parks, located mostly within the residential areas, all of which are highly managed and irrigated and (4) sub-natural areas that retain more of the native flora; these may be either urban parks that are lightly managed (if at all), or abandoned agricultural areas with traditional agricultural terraces (Ron 1985) such as in the Kos Wadi or olive groves (Ne'eman and Izhaki 1996) such as in the Valley of the Cross or in the Gazelle Valley (table 1). Locations and sampling time were chosen so as to reduce dependence among sites, thus preventing spatial autocorrelation and double counts between locations. Spatial autocorrelation can influence the null distribution of Pearson's R (referred to as R hereafter, and presented with the p value and the sample size n; Sokal and Rohlf 1995), leading to overestimating the number of degrees of freedom, and may therefore elevate the probability of a Type I error (Clifford *et al.* 1989). The PASSAGE software was used (available at http://www.passagesoftware.net/) to run a modified t-test for correlation, following Clifford *et al.* (1989). All correlation results were held highly significant after running this corrected test. To enable comparison among the four different area types, each of them included at least five sampling sites. Due to the high variation and heterogeneity among the different sub-natural areas, we sampled them such that they represented the range of sub-natural environments in the city of Jerusalem (i.e. open bathas and fields, semi-woodlands with scattered scrubs and more dense

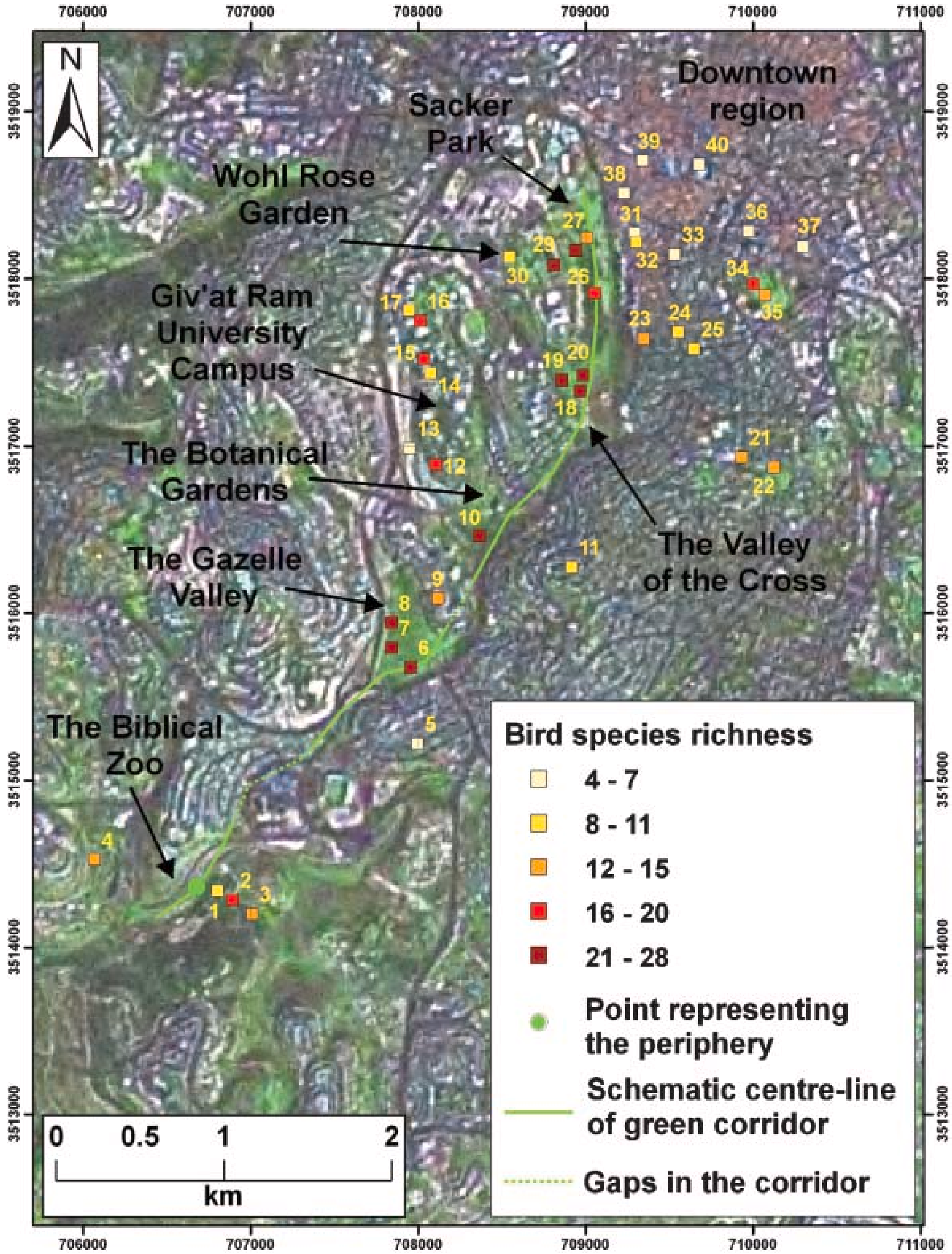

Figure 1. Map showing the sampling sites (with their numbers corresponding to those in table 1). The colour symbols represent the number of bird species sampled in each location over all surveys. The green line represents the green corridor (with gaps in the corridor shown as dotted lines), and the green circle represents the point at the periphery from which the distance to the periphery was calculated. In the background is a false colour image of Jerusalem from March 2002 (bands 7, 4 and 2 of Landsat).

woodlands, table 1). Due to differences in the degree of spatial heterogeneity among each of the four area types, their sample sizes were not equal. The sub-natural area type had the largest number of sampling sites (15), while the downtown area type

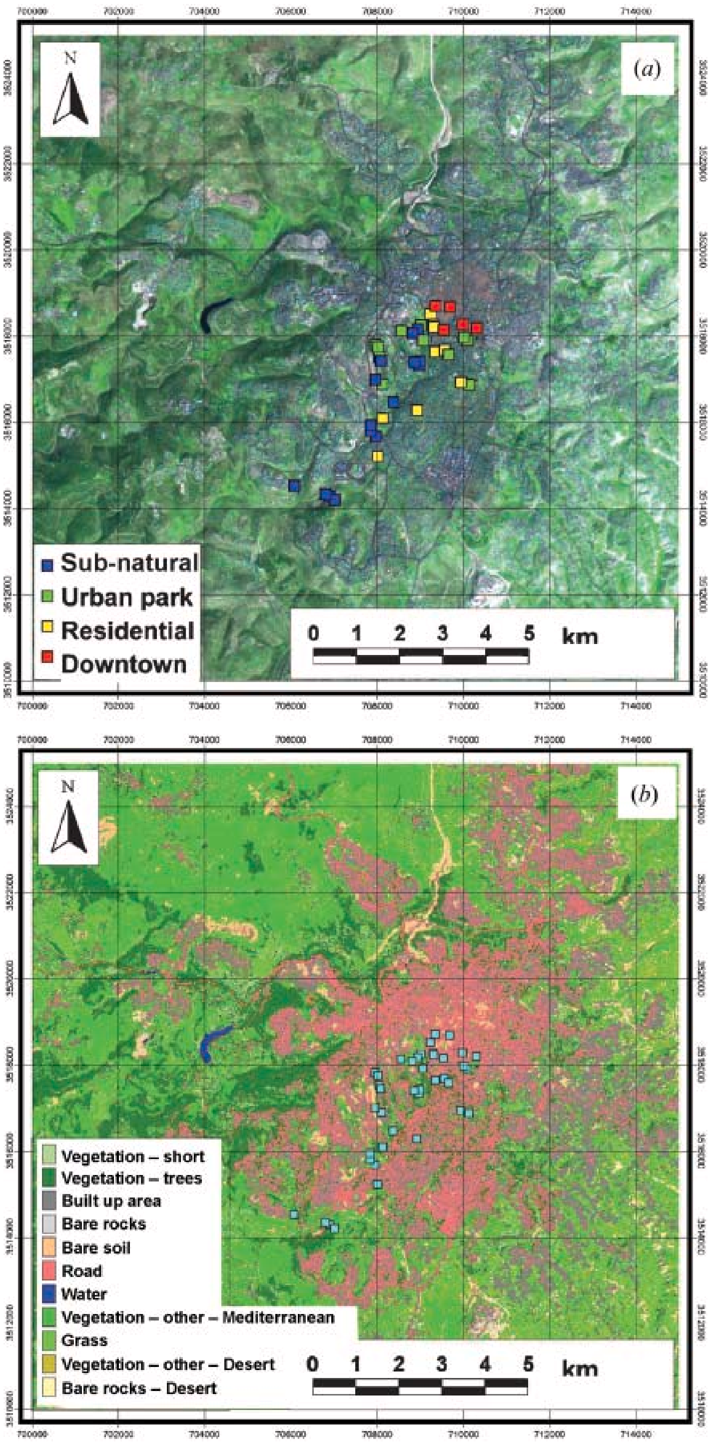

Figure 2. (*a*) False colour image of Jerusalem from March 2002 (Landsat bands 7, 4 and 2). The green colour represents vegetation and the purple shows built areas. Bird sampling locations are marked as squares for each of the four area types. (*b*) The classified land cover of Jerusalem based on the March 2002 image. Bird sampling locations are marked as squares.

Table 1. The study sites in Jerusalem. The numbers in column 2 correspond to those on figure 1. Plant species richness and green cover percentage (GCP) are given for the 15 sites that were sampled for vegetation. The number of sightings (an individual sighting of a bird or a flock of birds) and the minimum slope of the accumulation curve (as estimated by Sobs Mao Tao; Colwell 2005) represent two estimates of sampling effort.

Site name	No. on map in figure 1	Category	No. of times sampled	No. of sightings	Minimum slope of accumulation curve (species/sightings)	Bird species richness	Bird genus richness	Plant species richness	GCP from field data	Lawn dominated	Distance to periphery (m)	NDVI March 2002, 1 ha (Landsat based)	Roads and built-up areas (%), 1 ha (Landsat based)
Kos Wadi Forest	1	sub-natural	3*	37	0.11	11	11			−	130	0.236	14
Lower Kos Wadi	2	sub-natural	11	135	0.05	18	17	13	77	−	229	0.231	8
Upper Kos Wadi	3	sub-natural	10	97	0.04	14	14			−	369	0.254	5
Jerusalem Zoo	4	sub-natural	8	143	0.02	15	14			−	631	0.230	8
Isaac Sade St., Pat	5	residential	2*	17	0.17	6	6			−	1582	−0.001	77
Gazelle Valley Open	6	sub-natural	11	124	0.05	23	19			−	1845	0.386	1
Gazelle Valley Near	7	sub-natural	10	162	0.05	25	20	14	98	−	1856	0.348	0
Gazelle Valley Far	8	sub-natural	12	188	0.03	24	20			−	1973	0.363	0
Givat Mordechai	9	residential	9	119	0.04	14	14			−	2259	0.066	54
Hanna Herzog Garden	10	sub-natural	12	109	0.08	22	18	12	96	−	2708	0.267	4
Hamapilim St., Rasko	11	residential	2*	19	0.16	8	8			−	2954	−0.007	75
Campus dorms	12	urban park	12	177	0.02	20	19			−	2913	0.121	39
Campus (Thousand Forest)	13	sub-natural	2*	7	0.57	5	5			−	2927	0.077	42
Campus (Silberman)	14	sub-natural	2*	16	0.44	10	10			−	3388	0.132	27
Campus (Chemistry)	15	urban park	4*	45	0.15	16	16			+	3448	0.009	72
Campus Middle Lawn	16	urban park	7	111	0.03	17	17			+	3651	0.113	47
Campus—Berman	17	urban park	2*	20	0.1	8	8			+	3687	0.198	23
Valley of Cross Open	18	sub-natural	10	140	0.06	25	20			−	3759	0.147	8
Valley of Cross Close A	19	sub-natural	11	147	0.04	25	21			−	3746	0.311	1
Valley of Cross Deep B	20	sub-natural	12	188	0.04	28	23	16	74	−	3845	0.270	2
Pinsker St.	21	residential	6	98	0.02	14	13			−	4158	0.058	51
Rose Garden	22	urban park	10	183	0.01	15	15	13	65	+	4273	0.090	47
Iben Shaprut St.	23	residential	12	162	0	13	13	9	14	−	4236	0.019	63

Table 1. (*Continued.*)

Site name	No. on map in figure1	Category	No. of times sampled	No. of sightings	Minimum slope of accumulation curve (species/sightings)	Bird species richness	Bird genus richness	Plant species richness	GCP from field data	Lawn dominated	Distance to periphery (m)	NDVI March 2002, 1 ha (Landsat based)	Roads and built-up areas (%), 1 ha (Landsat based)
Ibn Ezra St.	24	residential	10	107	0.01	10	10	7	29	−	4403	0.029	59
Giraffe Garden	25	urban park	10	102	0.01	10	10	10	24	+	4389	0.033	64
Lower Saker Park	26	urban park	11	168	0.03	16	15	11	91	+	4284	0.313	3
Upper Saker Park	27	urban park	9	91	0.02	12	11			+	4541	0.215	11
Saker Park (Forest)	28	sub-natural	12	152	0.02	22	20	13	66	−	4438	0.269	8
Jerusalem Bird Observatory	29	sub-natural	11	187	0.02	24	19			−	4298	0.246	16
Rose Garden (Knesset)	30	urban park	2*	26	0.15	11	11			−	4217	0.178	21
Hamadregot St.	31	residential	4	74	0.01	7	7			−	4712	−0.054	68
Even Sapir St., Nahlaot	32	residential	8	103	0.02	9	9			−	4674	−0.037	65
HaGidem St.	33	downtown	2	28	0.04	5	5			−	4751	−0.048	81
Upper Independence Park	34	urban park	10	187	0.02	16	15	10	88	+	4916	0.268	9
Lower Independence Park	35	urban park	11	157	0.01	14	13	14	89	+	4916	0.231	7
Shamai St.	36	downtown	10	158	0	4	4	3	4	−	5133	−0.128	98
Shlomzion Hamalka St.	37	downtown	10	141	0	5	4	4	1	−	5280	−0.126	99
Behar St.	38	residential	12	134	0	6	6			−	4881	−0.091	69
Hashikma St.	39	downtown	2	25	0.08	5	5			−	5106	−0.116	95
Davidka Square	40	downtown	10	133	0	4	4	3	5	−	5272	−0.130	99

*Minimum slope of the accumulation curve $>$ 0.08 species/sightings.

had the smallest number of sites (five) (table 1). Owing to differences in the number of sampling sites in the four area types, we also examined the results obtained when combining the downtown and residential sites and compared them with the sub-natural sites (using a *t*-test; Sokal and Rohlf 1995).

2.3 *Bird surveys*

All sites were surveyed between August and October of 2003. This short sampling period enabled us to examine whether patterns can be detected when only part of the year is sampled, which is the case with many bird counts, such as the Christmas Bird Count in the USA that has been taking place during several days for 107 years now (http://www.audubon.org/bird/cbc/; Root 1988). Although we did not sample all seasons, clear patterns emerge (see Results). Bird point count surveys were carried out following Buckland *et al.* (2001, 2004). In order to produce a valid density estimate, each sampling point was divided into five concentric rings 10 m wide from 0 to 50 m. A sixth ring included distances over 50 m. All point counts were carried out by three of the authors (G.B., S.D., A.R.S.). Before beginning the survey at each point, we waited at the point for a 10-min 'calming period'. This was followed by a 10-min recording period, in which all birds seen or heard were recorded and the distance belt in which each bird was observed was marked. For birds seen in groups the flock size was recorded. Sampling points and hours of sampling (morning or afternoon) shifted among the surveyors to avoid sampler bias. As this season is also the onset of the autumn migration and bird diversity changes during the period, we ensured that during each surveyed week, all sampling points were visited once, thus ensuring comparable results between sites. Each point was surveyed both in the first 4 h following sunrise and during the 3 h prior to sunset, coinciding with peak bird activity. Altogether, 329 bird surveys were conducted.

Sampling effort was estimated for each of the 40 sites using a sample-based rarefaction curve (Colwell *et al.* 2004). The expected species accumulation curve was applied using the Sobs (Mao Tau) variable in the software EstimateS 7.5 (Colwell 2005). The accumulation curves were based on the accumulation of species per sighting (a sighting indicated either a single bird or a group (a flock of $\geqslant$2 birds). We compared the observed and the expected number of birds according to the accumulation curve for each site at two thresholds determined based on natural breaks that appeared on a scatter plot of sightings vs minimum slope (of the accumulation curve) at: (1) 70 sightings and (2) the point where the curve reaches a slope of 0.08 (species per sightings). The expected number of bird species at both thresholds was strongly correlated with the observed richness (r^2=0.98 and 0.97, respectively). We identified sites where sampling was insufficient as those where the accumulation curve did not reach a minimum slope of 0.08. These included eight out of the original 40 sites, and these eight were therefore excluded from some of the analyses. Saturation of the species accumulation curve was achieved mostly in the species-poor sites (figure 3).

2.4 *Bird richness and density calculations*

Bird richness was calculated at both the species and the genus levels. The latter was done to control for possible variation among the three observers in the degree of certainty in distinguishing between species based solely on audible records in a few genera (mainly among *Sylvia* species).

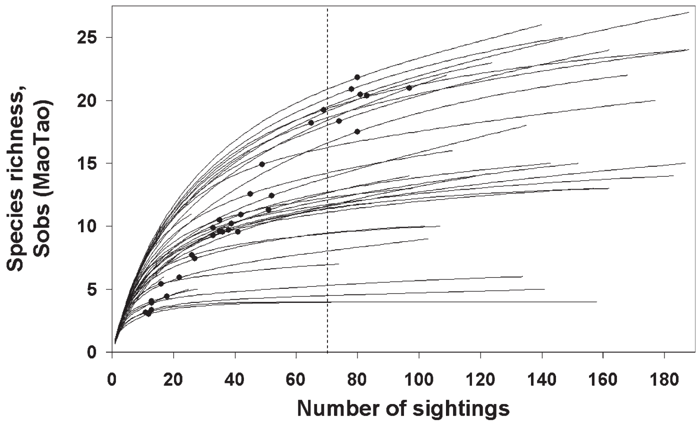

Figure 3. Accumulation curves for the 40 locations calculated using EstimateS 7.5 (Colwell 2005; see Methods). The vertical dashed line represents a threshold of 70 sightings per location, whereas the black dots represent threshold where the curve for each of the sampling locations reach a slope of 0.08 species per sighting.

The DISTANCE 4.1 program (Buckland *et al*. 2001, Thomas *et al*. 2004) was used to estimate densities (i.e. number of birds per ha). Species were grouped according to genus and data were stratified according to site and area type (Thomas *et al*. 2004). Before modelling, we truncated 10% of the largest bird distances recorded (Buckland *et al*. 2001). Data were modelled to produce a detection function for each genus. The half-normal model (both with and without the cosine and Hermite-polynomial series expansion) and the uniform (with and without the cosine and simple-polynomial series expansion) model were fitted to the data. The best model was selected using Akaike's Information Criterion (AIC) as calculated using DISTANCE.

2.5 *Field measurements of land cover*

Fifteen of the 40 sampling sites surveyed for birds were selected for additional field measurements of vegetation and land cover. These were chosen in sites with a bird sampling effort of at least 10 visits (table 1). Land and canopy cover were sampled in the field between February and March 2004, when the area is green and annuals can be identified. At each of the 15 sites, starting from the same point in which the bird point counts were performed, we marked eight transects (every 45°), each 50 m long. We recorded the length of all land-cover features along each transect using a measurement tape. The features measured included trees, perennials, annuals, grass lawns, bare ground, rock and asphalt. Because land and canopy cover were measured, values greater than 100% could be attained. We normalized land and canopy cover by total cover to calculate the percentage cover in each site. The green cover percentage index (GCP) was then calculated, which is the total cover for trees, perennials and annuals in all eight transects within each site combined.

2.6 *Satellite images and remote sensing application*

Two Landsat 7 satellite images were used (WRS-2, path 174, row 38; see Cohen and Goward (2004) for a review on Landsat and ecological studies). The images were selected so that they correspond to both the dry season (August, when the birds were sampled in the field) and the time of year of highest productivity (spring, when the plants were sampled in the field). One image was acquired in the summer (7 August 1999; processing level: L1G) and the other in the early spring (8 March 2002; processing level: Geo Cover orthorectified, as part of NASA's Geocover dataset; Tucker *et al*. 2004). We used images from more than one season to also enable us to differentiate between natural vegetation and irrigated lawns (where only the latter remain green in the summer), as done in irrigation studies that use remote sensing (Thenkabail *et al*. 2006). We ensured that there was no significant change in the amount of built or green area in the sampled areas between 1999 (year of the first image) and 2004 (year of the bird sampling). The images were downloaded from http://glcfapp.umiacs.umd.edu:8080/glcf/esdi. The pre-processing stages of the satellite images included these steps:

1. Band sharpening, applying the most commonly used algorithm known as Intensity–Hue–Saturation (IHS; see Vrabel 1996, Park and Kang 2004). This was done to improve mapping accuracy of the Landsat image, as the characteristic scale of urban reflectance is usually between 10 and 20 m (Small 2003).
2. Correction of each of the bands for shading effects caused by the topography following Smith *et al*. (1980), using a Digital Elevation Model (DEM) obtained from the Survey of Israel (Hall *et al*. 1999).
3. Rectification of the satellite images to the Israel New Grid (Israel Transverse Mercator; Mugnier 2000) coordinate system, based on 14 ground control points (GCPs) identified on a colour orthophoto (1 m resolution), by applying a first-order polynomial transformation (root mean square error of 7.7 m, which is equivalent to half the pixel size of the panchromatic band).

Following this process, the Normalized Difference Vegetation Index (NDVI; Tucker 1979) was calculated for August 1999 and March 2002, as follows:

$$\mathrm{NDVI} = \frac{(B_4 - B_3)}{(B_4 - B_3)} \tag{1}$$

where B_4 is the near-infrared band digital number (DN) value and B_3 is the red band DN value. NDVI is unaffected by band sharpening since the relative contributions of the near-infrared and red bands remain unchanged by the fusion method (Wiemker *et al*. 1998). The NDVI maps were further low-pass filtered using average kernels (Clark Labs 2002) to analyse the patterns of NDVI over different spatial resolutions of 1 pixel (0.02 ha $=14.25\times14.25$ m), 3×3 pixels, 5×5 pixels, 7×7 pixels (1 ha), 9×9 pixels, 19×19 pixels, 23×23 pixels, 27×27 pixels (14.8 ha), 41×41 pixels, 55×55 pixels, 69×69 pixels and up to a coarse scale of 81×81 pixels (133 ha). This was done to analyse the effect of vegetation cover of the surrounding area on bird richness and to identify the scale in which the relationships are strongest.

Next, we generated a classified map of the urban and open space matrix of the city of Jerusalem and its environs based on the March 2002 image. This was done by identifying 11 classes of regions of interest on the satellite image: built-up areas, roads, water bodies, bare soil, two types of bare rocks (in desert or Mediterranean areas), urban grass lawns, and four types of natural vegetation. As urban vegetation cover can be reliably estimated using linear spectral unmixing (Small 2001), we applied a supervised unmixing mapping method implemented in Idrisi 32 called probability guided linear spectral unmixing (Clark Labs 2002). Eleven maps were produced representing the percentage cover of each one of the 11 classes in the image. The per cent cover of grass was mapped based on both summer and spring images, as irrigated grass remains green and shows high NDVI values year-round. These maps were then combined into a final classified map of Jerusalem using a multi-dimensional choice procedure in which each pixel was assigned to the class that had the highest percentage in that pixel (figure 2). The per cent cover maps were further low-pass filtered (Clark Labs 2002) to analyse land-cover patterns at various spatial resolutions, as described for the NDVI above. For the correlation analyses with bird richness, we grouped the buildings and roads to one category representing the built-up area, and the three sub-categories of natural Mediterranean vegetation to one category that we termed natural Mediterranean vegetation. These two grouped categories contain the majority of percentage land cover within the city of Jerusalem. Both variables were arcsine transformed to meet with normality assumptions. In addition, we calculated Shannon's diversity index (Shannon and Weaver 1949, see also Alberti *et al.* 2000) applying it to the land-cover classes (Clark Labs 2002). Initially the data were analysed at a scale of 7×7 pixels (1 ha, closely corresponding to the area that was sampled in the field of $0.8\,\text{ha} = \pi \times 50^2\,\text{m}$).

We ran a multiple regression analysis for bird richness as the dependent variable with independent variables derived from three sources: (1) remote sensing—the 11 types of land cover and the NDVI index; (2) field land cover measurements—green cover percentage and plant species richness; (3) geographical—distance from the city periphery. Data distribution was tested and was found to meet with the assumptions required for general linear models, which were then applied. The results of the multiple regression include the value F (of F-test for the overall fit of the model), the degrees of freedom d.f., and the relative predictive importance of the independent variables (Sokal and Rohlf 1995). Variables that were found to be not significant in the regression model are noted as 'NS'. To test for collinearity, the variance inflation estimator (VIF) estimate for collinearity was used, which represents $1/(1-R_i^2)$, or the amount that the variance of the ith regression coefficient is inflated due to collinearity (Philippi 1993), as calculated by JMPIN 5.1 statistical software (2003, SAS Institute, Cary, NC, USA). We made sure that the VIF was low for all variables in the model. When collinear variables showed values of VIF higher than 10, we saved only one of them in the model, following Philippi (1993).

3. Results

3.1 *Factors shaping the patterns of bird distribution*

Overall, during the sampling period, 40 bird species were recorded. These belong to 30 bird genera. Using the spring image, mean NDVI in a 1-ha area surrounding each sampling point was positively correlated with bird species richness ($R=0.86$, $p<0.001$, $n=32$). When the seven urban park sites dominated by irrigated lawn were excluded from the analysis, the correlation became even stronger at both the species level ($R=0.92$, $p<0.001$, $n=25$) and the genus level (table 2). When the summer image was used, NDVI values were less strongly, though significantly, correlated with bird species richness ($R=0.36$, $p=0.043$, $n=32$). When the irrigated lawn areas were excluded from the summer image the relationship became stronger both at the species level ($R=0.64$, $p<0.001$, $n=25$) and the genus level (see table 2). The relationship between NDVI and species richness showed a hump-shaped pattern when examined over increasing spatial scales (see figure 4). R^2 between bird richness and NDVI increased from the finest scale (an area of 0.02 ha surrounding each of

Table 2. Summary of the correlation values between different remote sensing indices and (i) bird species richness (BSPR) and (ii) bird genus richness (BGR). Three sample sizes (number of locations sampled) are compared: n=40 (including all sampled locations), n=32 (excluding locations where the accumulation curve did not reach a minimum slope of 0.08 species per sightings) and n=25 (excluding also locations with irrigated lawns).

		NDVI						
	Distance to city periphery	August 1999 (1 ha)	March 2002 (0.02 ha)	March 2002 (1 ha)	March 2002 (14.8 ha)	Correlation between bird richness and percentage road/built area (March 2002) (1 ha)	Correlation between bird richness and percentage natural vegetation (March 2002) (1 ha)	Correlation between bird richness and Shannon's landscape diversity index (March 2002) (1 ha)
BSPR, n=40	−0.31	0.33*	0.69†	0.79†	0.87†	−0.77†	0.76†	0.29
BSPR, n=32	−0.49†	0.36*	0.82†	0.86†	0.91†	−0.86†	0.84†	0.44*
BSPR, n=25	−0.50*	0.64†	0.88†	0.91†	0.93†	−0.92†	0.91†	0.54†
BGR, n=40	−0.34*	0.39*	0.70†	0.79†	0.87†	−0.77†	0.74†	0.35*
BGR, n=32	−0.53†	0.43*	0.84†	0.87†	0.91†	−0.86†	0.82†	0.51*
BGR, n=25	−0.55†	0.68†	0.89†	0.92†	0.93†	−0.93†	0.92†	0.60†

*$p<0.05$, †$p<0.01$.

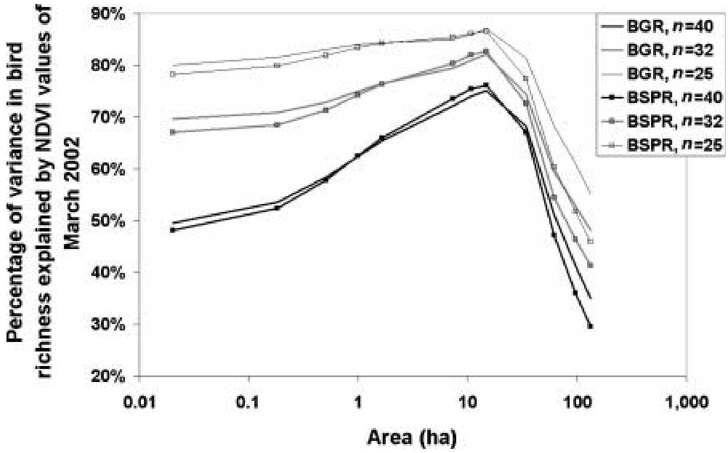

Figure 4. The percentage of the variance in bird species richness (BSPR) and bird genus richness (BGR) explained by spring NDVI values, as a function of the spatial scale (area around the sampling location). Three sample sizes (numbers of locations) are compared: $n=40$ (including all sampled locations), $n=32$ (excluding locations where the accumulation curve did not reach a minimum slope of 0.08 species per sighting), and $n=25$ (excluding also those locations with irrigated lawns).

the sampling points; $R^2=0.67$, $p<0.001$, $n=32$), up to a maximum value at a scale of 14.8 ha ($R^2=0.83$, $p<0.001$, $n=32$) and then declined again. The same pattern was found at the genus level (see figure 4 and table 2).

Spectral unmixing of land cover using remote sensing tools yielded several efficient individual explanatory variables; the most prominent being those with an adverse effect such as road cover ($R=-0.84$, $p<0.001$, $n=32$) and built-up area cover ($R=-0.76$, $p<0.001$, $n=32$). Both tree and short vegetation cover were positively correlated with bird species richness ($R=0.69$, $p<0.001$, $n=32$ and $R=0.73$, $p<0.001$, $n=32$, respectively). The grouped land-cover variables represented species richness in the most effective way ($R=-0.85$, $p<0.001$, $n=32$ for built-up and road cover, and $R=0.85$, $p<0.001$, $n=32$ for Mediterranean vegetation cover). Shannon's land-cover habitat diversity index calculated based on the classification procedure also showed a significant correlation with the bird species and genera richness in each location ($R=0.44$, $p=0.001$, $n=32$ and $R=0.51$, $p=0.001$, $n=32$, respectively).

Distance of sites from the city periphery was negatively correlated with bird species richness ($R=-0.49$, $p=0.05$, $n=32$). Similar results were found at the genus level (see table 2). Distance from the periphery and March NDVI values (calculated at 1 ha scale around the point sampling) were negatively correlated ($R=-0.49$, $p=0.001$, $n=40$).

A stepwise multiple regression explained 84% of the variation in bird species richness ($F=47.2$, d.f.$=31$, $p<0.001$). The variables incorporated into the model included the percentage of road and built area (unstandardized $\beta=-8.4$ $t=-4.8$, $p=0.001$), percentage of natural vegetation ($\beta=12.8$, $t=4.6$, $p<0.001$) and distance of sites to city periphery ($\beta=0.01$, $t=2.4$, $p=0.025$) while excluding NDVI (March, 1 ha scale) ($\beta=0.2$, $t=0.6$, NS). In this model, all VIF values were much lower than 10 (the highest being 3.1). NDVI was excluded from the regression model due to its strong negative correlation with the percentage of road and built area ($R=-0.97$, $p=0.0001$, $n=40$), which led to high VIF values in the regression model.

3.2 *Field measurements of plants*

Plant species richness as measured in the field declined from the downtown towards the sub-natural sites (table 1). Plant richness was positively correlated with bird richness ($R=0.87$, $p=0.0001$, $n=15$). The site with the lowest number of plant species also had the lowest number of bird species and vice versa for the site with the highest number of plant species (table 1). Mean green cover percentage (GCP) increased from 3.3% in the downtown sites to 82.2% in the sub-natural sites. The main increase was between the residential areas (21.6%) to the urban park sites (73%) (table 3). GCP was positively correlated with bird species richness ($R=0.80$, $p=0.0001$, $n=15$). Perennial cover was positively correlated with bird species richness ($R=0.61$, $p=0.02$, $n=15$). However, tree cover, as measured in the field, was not significantly correlated with bird species richness ($R=0.29$, $p=0.29$, $n=15$). Asphalt cover showed a negative correlation with bird species richness ($R=-0.81$, $p=0.0001$, $n=15$) and with bird genus richness. Average NDVI values based on the spring image were strongly correlated with both GCP ($R=0.97$, $p=0.0001$, $n=15$) and with plant species richness ($R=0.90$, $p=0.0001$, $n=15$). A multiple regression between bird species richness as the dependent variable and perennial cover and plant species richness as predictors had an R^2 of 0.79 ($p<0.001$, $n=15$). All the above relationships persisted when examined for bird genus richness.

3.3 *Bird diversity patterns*

All species observed but two, the rose-ringed parakeet (*Psittacula krameri*) and the laughing dove (*Streptopelia senegalensis*), are native to Israel (Shirihai 1996). An ANOVA showed a significant difference in bird species richness between the four area types ($F=36.04$, $p<0.001$, d.f.$=31$). Average bird species and genus richness increased from the downtown to the residential areas and urban parks, and reached a peak in the sub-natural areas (table 3). The richest site, in which 28 bird species were recorded, was located in a sub-natural area, while the two poorest sites, where only four bird species each were recorded, were located in the downtown area (table 1). These species were the house sparrow (*Passer domesticus*), hooded crow (*Corvus corone*), rock dove (*Columba livia*), and laughing dove (*Streptopelia senegalensis*). Average bird species richness within the sub-natural area types was 22.1 ($n=12$) compared with an average richness of 8 ($n=12$) in the residential and downtown area types combined ($z=-4.1$, $p<0.001$, $n=24$). All bird species sampled in the downtown, residential and urban park areas of the city were also sampled in the sub-natural areas, whereas 11 species were restricted to the sub-natural areas.

All 30 genera recorded throughout the survey were present in at least one sub-natural site, of which seven were unique to sub-natural areas and were not found in residential, commercial or urban park sites. The composition of species with increasing urbanization as quantified based on the NDVI showed an uneven distribution of species (figure 5). Some species peaked in residential areas (e.g. Palestine sunbird (*Nectarinia osea*) and *Streptopelia* sp. with an abundance of 50.8 and 354.1 individuals per km^2, respectively), some peaked in urban parks (e.g. Syrian woodpecker (*Dendrocopos syriacus*) and *Corvus* sp.; 10.4 and 247.5 individuals per km^2, respectively), and others in the sub-natural areas (e.g. graceful prinia (*Prinia gracilis*) and *Sylvia* sp.; 10.2 and 100 individuals per km^2, respectively). Some species showed dual peaks, at both the residential and sub-natural areas (e.g. spectacled bulbul (*Pycnonotus xanthopygos*) and shrikes (*Lanius* sp.); 104.3/107.4 and 6.3/8.6 individuals

Table 3. Species distribution in each of four area types sampled in Jerusalem, including downtown locations, residential, urban parks and sub-natural areas. The table includes only those 32 sites where the accumulation curve reached a minimum slope of 0.08 species per sightings.

Area type	Mean (±SD) bird species richness	Mean (±SD) bird genus richness	Mean (±SD) plant species richness	Mean (±SD) green cover percentage (GCP)	Mean distance to periphery (m)	Mean NDVI	Mean proportion road and built-up area
Downtown (n=5)	4.6±0.6	4.4±0.6	3.3±0.6 (n=3)	3.2±2.1 (n=3)	5108.4	−0.110	0.47
Residential (n=7)	10.4±3.3	10.3±3.1	8.0±1.4 (n=2)	21.6±11.1 (n=2)	4189.1	−0.002	0.31
Urban park (n=8)	15.0±3.1	14.4±3.0	11.3±1.9 (n=4)	73.0±32.4 (n=4)	4235.3	0.173	0.14
Sub-natural (n=12)	22.1±4.3	18.8±2.7	13.6±1.5 (n=5)	82.2±14.1 (n=5)	2474.7	0.277	0.025

June 2008

Figure 5. A presence (black square) and absence (white square) summary of the genera sampled at least once per site (the figure is given at the genus level). The sites are ordered according to their decreasing spring NDVI value (for an area of 14.8 ha), whereas the genera are ordered according to the number of sites in which they are present. *Sylvia sp. include *Sylvia communis*, *Sylvia curruca*, *Sylvia hortensis*, *Sylvia melanocephala* and *Sylvia atricapilla*.

Table 4. Average bird densities (individuals/km^2), calculated for the four area types (downtown, residential, urban parks, and sub-natural areas).

Latin name	Downtown	Residential	Urban parks	Sub-natural
Passer domesticus	958.9	561.1	444.2	213.2
Columba livia	770.2	365.4	99.5	11.4
Streptopelia sp.	79.4	354.1	58.4	12.4
Corvus sp.	78.2	55.7	247.5	25.6
Pycnonotus xanthopygos	2.2	104.3	63.9	107.4
Nectarinia osea		50.8	16.4	27.3
Turdus merula		20.7	46.5	38.8
Merops apiaster		9.7	0.6	2.9
Carduelis sp.		9.2	11.9	43.3
Dendrocopos syriacus		9.1	10.4	7.9
Lanius sp.		6.3	1.2	8.6
Garrulus glandarius		5.8	25.3	7.2
Falco sp.		4.9	0.9	1.2
Parus major		3.9	19.6	24.6
Emberiza hortulana		1.0	0.2	1.5
Psittacula krameri		0.5	1.3	14.9
Sylvia sp.		0.4	8.5	100.0
Prinia gracilis		0.4	7.0	10.2
Accipiter nisus		0.4	0.1	0.3
Upupa epops			2.1	0.1
Hirundo rustica			0.9	0.3
Phylloscopus trochilus			0.7	5.3
Hippolais pallida			0.4	2.9
Apus apus				4.4
Saxicola rubetra				0.3
Muscicapa striata				3.5
Phoenicurus phoenicurus				11.3
Alectoris chukar				7.9
Vanellus spinosus				0.4
Halcyon smyrnensis				2.2

per km^2, respectively). All genera recorded in the downtown area showed very high densities there (e.g. house sparrow and rock dove with 959 and 770 individuals per km^2, respectively). These species also had high densities in residential and urban parks areas, yet declined sharply in the sub-natural areas (table 4). On the contrary, species restricted to sub-natural sites were found in relatively low densities (mean of 4.3 individuals per km^2, n=7) with an exception of *Sylvia* sp., which had high local densities (mean 100 individuals per km^2) in sub-natural sites.

4. Discussion

4.1 *Relationship between remote sensing and urban biodiversity*

In this study, we used Landsat-derived NDVI estimates and land-cover maps generated by spectral unmixing to study urban patterns of avian biodiversity and compare them with field sampled per cent plant cover and richness. NDVI correlates strongly with plant biomass and with net primary productivity (NPP), the difference between carbon fixed by photosynthesis and the carbon lost to autotrophic respiration (Kerr and Ostrovsky 2003, Evans *et al.* 2005). As such, it is one of the best available estimates for energy in a system, as well as of the system's productivity. The relationship between energy, productivity, and species richness is an extensively studied topic in ecology and has been tested at multiple spatial scales and ecological systems (Evans *et al.* 2005).

In recent years an increasing number of studies are examining the relationships between NDVI as an estimate of energy and richness patterns in various groups. However, up to now most studies have focused on non-urban, rather than urban landscapes. In more natural systems, various relationships have been reported in recent studies between NDVI and species richness, including a positive (Hurlbert and Haskell 2003, Bailey *et al.* 2004, Seto *et al.* 2004, Gillespie 2005, Levin *et al.* 2007), a hump-shaped (Bailey *et al.* 2004, Fairbanks and McGwire 2004, Seto *et al.* 2004) and a negative relationship (Oindo and Skidmore 2002). Because the city shows strong contrast between built-up and sub-natural areas, we expected remote sensing tools to perform successfully in predicting alpha diversity (richness) patterns. In our study, we found a strong negative relationship between the per cent cover of built-up areas and bird richness (see also Alberti *et al*. 2000), mirrored by a positive relationship between NDVI and bird richness. The strong linear relationships (rather than hump-shaped) that we found between NDVI and species richness may be due to the strong limiting factor imposed by built-up areas on species richness; thus, very high levels of productivity in which species richness declines are not reached in the city. However, in a highly human-modified environment, this relationship may be altered by various factors such as irrigated lawns. These sites, primarily urban parks, are highly managed and irrigated throughout the year and thus maintain high NDVI values, but support only few species. Using remote sensing tools, these habitats can be differentiated from sub-natural areas by comparing images from the spring vs the summer, as we did here. While the images taken in both the spring and the summer were useful, the NDVI values calculated based on the spring images were more strongly correlated with bird and plant richness as sampled in the field. This is due to the fact that in the Mediterranean climate region (Hobbs *et al.* 1995), where the city of Jerusalem is located, most of the annuals disappear during the summer and many of the shrubs become less green compared with spring (Schmidt and Karnieli 2000, Shoshany and Svoray 2002). This leads to lower NDVI values in sub-natural areas within the city during the summer. Indeed, when plotting the NDVI values from both seasons (August and March), most sites with high spring NDVI values show lower values in the summer.

Field sampling of land-cover categories corresponded well with the remote sensing mapping of urban land cover. Areas with higher paved and built area cover hold a lower number of both plants and bird species (see also Alberti *et al*. 2000). Field-based estimates of land cover and of species richness are both costly and resource consuming, especially when carried out over large areas and at multiple spatial scales. Remote sensing can easily be applied and enables one to monitor large areas at low expenses. It is therefore a useful tool for conservation scientists and for decision makers (Diamond 1988, Fairbanks and McGwire 2004). The high correlation between bird richness and NDVI suggests that NDVI may serve as an efficient tool for monitoring and estimating bird richness and plant cover and richness within the city (see also Nichol and Lee 2005).

Comparison of different spatial scales can provide us with information regarding the optimal scale for studying the relationship between species richness and the environmental variables affecting it (Savard *et al.* 2000, Jokimaki and Kaisanlahti-Jokimaki 2003), as well as the natural and human-related processes shaping the patterns (Willis and Whittaker 2002, Pautasso 2007). We examined the effect of spatial scale on the relationship between NDVI and bird richness by calculating NDVI values at various area sizes and identifying the scale where the maximum correlation between bird richness and NDVI occurs. This scale was termed by Hostetler and Holling (2000) as the 'best prediction area'. The highest correlation between NDVI and bird richness was found at an area of 14.8 ha. This corresponds with the best prediction area seen for the majority of North American bird species sampled in summer surveys across several urban sites (Hostetler and Holling 2000). Our results suggest that analyses at different scales can lead to different conclusions as to the spatial scale and size of green area needed to maintain highest native biodiversity in the city (Pautasso 2007). As the correlation between NDVI and bird species richness was highest at the spatial scale of ~15 ha (equivalent to a pixel size ~390 m^2), it appears that satellite images with a moderate spatial resolution (250–500 m) may be also efficient in urban studies of avian richness patterns. An example for such a sensor is MODIS, whose products are freely available over the Internet, and from which both NDVI and land-cover layers and maps can be created (Huete *et al.* 2002, Wessels *et al.* 2004).

4.2 *Biodiversity along urban gradients*

Previous studies have shown that downtown areas, as expected, are species poor environment for birds (Blair 1996, Clergeau *et al.* 1998, Blair 1999, Melles *et al.* 2003, Crooks *et al.* 2004). In this study, we expand this finding and clearly show a gradient of increasing bird richness as one moves from the downtown towards sub-natural areas within the city. Remote sensing estimates of NDVI and spectral unmixing of land-cover classes support our field work results, showing clear relationships between vegetation cover estimations and bird richness, where greener and less built up areas show higher richness. Richness of birds in the city may be affected by the size of the sub-natural habitat patches, corresponding with the species–area relationship, according to which species richness increases with increasing area size (Arrhenius 1921, Fernandez-Juricic 2000, Lomolino 2000, Donnelly and Marzluff 2004). However, in our case, all sub-natural sites within the city, even the smaller ones, maintain relatively high bird richness. This could result from the relatively high connectivity among sub-natural sites within the city. The sub-natural sites are all located along the green corridor, which is relatively continuous from the city periphery inwards (figure 1). This fact may enable birds to move between sites along the green corridor (Clergeau *et al.* 1998). Birds can potentially move large distances with relative ease, especially when compared to other organisms (e.g. small mammals and flightless insects). Therefore, their locations are likely dependent on active habitat selection (Flink and Seams 1993, Collinge 1996) in the highly heterogeneous urban environment.

When comparing the local community composition among the different areas within the city, we find an uneven distribution of birds with a biotic homogenization effect seen with increasing urbanization (Faeth *et al.* 2005, Clergeau *et al.* 2006, McKinney 2006, Olden 2006, Shochat *et al.* 2006). In our study, a small number of species were extremely widespread, observed along the entire range of the urban gradient (figure 5). These urban exploiters, as termed by Blair (1996), included the hooded crow (*Corvus corone*), the house sparrow (*Passer domesticus*), the rock dove (*Columba livia)* and two species of the genus *Streptopelia*, which were found in 97.5%, 95%, 92.5%, 85% and 85% of the 40 sites, respectively. These urban exploiters are well adapted to the urban environment (Blair 1996, Marzluff 2001, Sorace 2002, Crooks *et al.* 2004, Kark *et al.* 2007) and many of them have established breeding populations outside their native range as alien species (Long 1981, Lever 1987). Kark *et al.* (2007) have recently showed that being successful in more vs less urbanized environments in the city is not necessarily a factor of brain size or behavioural innovations of a bird species, but rather is related to a combination of traits, including diet, degree of sociality, sedentariness, range expansion and preferred nesting sites.

This study shows strong relationships between NDVI and bird richness although only a single season was sampled. The species accumulation curves show saturation in the number of species for the downtown areas, whereas saturation in the accumulation curve for the more species rich sites is not reached. Thus, additional sampling effort over a longer season and comparison of different seasons will be interesting and may lead to even larger differences between the species-poor downtown and richer sub-natural areas. If patterns seen over a short sampling season are similar to those in the longer seasons, this may suggest that for monitoring purposes, a short sampling period may suffice, enabling to wisely invest our limited resources and to sample additional areas.

4.3 *Implications for planning*

Whereas in the past planners, city municipalities, environmental organizations and ecologists were focused on conservation of wilderness areas, awareness of the importance of natural areas within cities, where most of the human population resides, is increasing (Breuste 2004, Liven-Schulman *et al.* 2004, Bryant 2006). This is also true for the city of Jerusalem, where there is an active environmental community of local people from various organizations, that act to preserve the open spaces still remaining within and around Jerusalem (e.g. see http://www.jbo.org.il/Eng%20index.htm; http://www.sustainable-jerusalem.org/old_site/jerusaleme/jerusaleme.html).This study emphasizes the potential of applying remote sensing tools in conservation planning and decision making in the urban landscape. These tools can be used at different spatial scales, are highly repeatable, and are useful for monitoring purposes, as data can easily be compared over time. The recent advancement in the availability of high resolution satellite images (Fritz 1999) and of global Landsat images that are now freely available on the web (NASA's Geocover dataset; Tucker *et al.* 2004) is now enabling us to directly estimate productivity using vegetation indices, and to examine the relationships between these estimates and biodiversity patterns (Turner *et al.* 2003). The current cost of about 600 US$ for a Landsat image makes it very affordable, and for small areas high resolution images (e.g. QuickBird) can be obtained at $1500. Whereas the ecological field work in this study involved five people for 3 months, the processing of the satellite imagery was done by just one person in several days. While remote sensing is not able to replace field work and cannot identify single bird species, rarity and composition, we suggest that given the useful products of such images, investment in image purchase and analysis will be highly cost-effective. In this regard even the availability of Google Earth (http://earth.google.com; Lisle 2006) is allowing unprecedented views of urban areas around the globe.

5. Conclusions

In this work, we found that spatial patterns derived from remotely sensed data, both NDVI and spectral unmixing of urban land cover, can be very useful in explaining bird richness in the city. The results obtained from remote sensing derived data were on a par with field-derived data. In such cases, cost-effective considerations make the use of remotely sensed data advantageous. In recent years, there has been an increase in the use of hyper-spectral images to study urban areas and other human-dominated landscapes (e.g. Ben-Dor *et al.* 2001, Benediktsson *et al.* 2005, Levin *et al.* 2007), and we expect that this development will allow better identification of species-rich habitats. While remote sensing approaches are very promising, field surveys are obviously crucial for identifying the community structure, for identifying rare species, for examining alien species and for understanding the underlying factors that lead to the patterns. We believe that a combination of remote sensing tools and fieldwork is essential. Comparing the patterns in other taxa that respond to different spatial scales and their relation to NDVI would be interesting.

In the city of Jerusalem, the crucial factor shaping bird diversity was not the distance to the city edge but rather the type of environment. Those species that occur in the downtown area are also common in other parts of the city, and are well adapted to human environments, some of them becoming highly successful invaders in many other parts of the world. Thus, the richness and uniqueness of birds in the city largely depends on how much semi-natural habitat is left for biodiversity to persist in the city. It is clear that sub-natural areas within the city, even when small in size, can maintain high bird richness, much higher than built-up areas. Eleven of the 40 species sampled in the city only occurred in sub-natural areas. These sub-natural areas represent a wide range of different habitats, including abandoned olive groves, deserted agricultural orchards and open Mediterranean bathas, generally retaining more of the native flora. Most of them are only lightly managed, if at all, and human impact and activity in them is much lower than in other parts of the city. Thus, in order to enable the persistence of high native bird diversity in an urban landscape one should maintain patches with semi-natural vegetation, which may maintain high levels of bird diversity even when distances from the city periphery are relatively high. Remote sensing tools may be applied to examine changes in land cover, in NDVI and in the distribution of sub-natural and other environments in the city. Based on our study, these tools may be successfully used as part of our attempts to study and predict biodiversity patterns in the city both today and in the future.

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