

ACTA UNIVERSITATIS CAROLINAE

European Journal of Environmental Sciences

VOLUME **11** / NUMBER **1**
2021



CHARLES UNIVERSITY
KAROLINUM PRESS

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ISSN 1805-0174 (Print)
ISSN 2336-1964 (Online)

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INFLUENCE OF INTRASPECIFIC COMPETITION FOR FOOD ON THE BODYWEIGHT OF THE ADULT APHIDOPHAGOUS LADYBIRD, *COCCINELLA TRANSVERSALIS*

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ABSTRACT

Aggregation of conspecific predators sharing a common prey, influences their bodyweights. We investigated the influence of intraspecific competition of adult ladybirds of *Coccinella transversalis* Fabricius on their bodyweight feeding on rusty plum aphid, *Hysteroneura setariae* (Thomas). Adult males and females consumed a significantly greater number of aphids with increase in predator-density, however, the aphid-consumption per predator declined with this increase. The weight gain per predator also decreased linearly with increase in the density of both male and female predators. This indicates that the weight-gain of the predator is a function of the prey consumed. The searching efficiency decreased with increase in predator density due to mutual interference. The mutual interference constants for adult male and female ladybirds were -0.419 and -0.546 , respectively. The females consumed a greater number of aphids than males. The killing power of the ladybird denoted by the k -value increased curvilinearly with increase in predator density. We conclude that prey consumption is a function of body size and that the offspring of those that aggregate at low densities in prey-rich habitats develop into large adults.

Keywords: conspecific predator; *Coccinella transversalis*; intraspecific competition; ladybirds; numerical response

Introduction

Knowledge of the predator-prey interactions of predatory ladybirds (Coccinellidae: Coleoptera) is important for understanding their effectiveness in the biocontrol of aphids. Quantitative estimates of ladybirds' searching efficiency and prey consumption at varying prey-densities indicate their potential as biocontrol agents (Bayoumy 2011; Bayoumy and Michaud 2012). This predator's functional response to the changes in prey density indicates density-dependent prey consumption (Holling 1959). However, the effect of predator density on prey density may also help predict biocontrol outcomes, estimate the effect of intraspecific competition and interferences among ladybirds. The density-dependent predator-prey dynamics is described by numerous models (Pervez et al. 2018), of which the classical Nicholson and Bailey (1935) model defines "area of discovery", as a crucial parameter determining the searching efficiency of a predator. An inductive model (Hassell and Varley 1969) including the mutual interference constant (Hassell 1971; Bayoumy et al. 2014), further simplifies this model and indicates that the predator's searching efficiency declines with increase in its density. These models advocate predation to be a function of both prey- and predator-dependent processes and account for the effect of mutual interference on prey consumption. This interference alters ladybird's foraging success or may compensate for the decline in foraging activity due to the time required for digestion at high prey densities (Papanikolaou et al. 2016). Kindlmann and Dixon (1993) questioned the biocontrol potential of aphidophagous ladybirds stating that even optimal foraging and laying of eggs may only result in a slight reduction of aphid abundance. Furthermore, the adults

should maximize their fitness by deciding whether to stay in or leave an aphid-patch (Kindlmann and Dixon 2010). In addition, greater generation-time ratio of ladybirds makes them slow developers, thereby impeding the top-down regulation of aphid abundance (Kindlmann and Dixon 1999, 2001, 2015). Kindlmann et al. (2020) further concluded that it is generation-time ratio rather than voracity that drives the dynamics of insect-natural enemy systems, particularly aphid-ladybird system.

Predaceous ladybirds (Coleoptera: Coccinellidae) are potential biological control agents, as they prey upon numerous coccid and aphid pests (Hodek et al. 2012; Omkar and Pervez 2016; Pervez et al. 2020). They switch from extensive search to intensive search after capturing a prey (Pervez and Yadav 2018). Complex plant morphology further modifies intensive search (Legrand and Barbosa 2003). Mutual interactions impede their consumption of prey and searching efficiency (Omkar and Pervez 2004a; Bayoumy and Michaud 2012). Their searching efficiency and incidence of mutual interference might be dependent on the type of prey (Al-Deghairi et al. 2014). These coccinellid predators may switch from a rare stage of prey to an abundant stage of prey (Fathipour et al. 2020) thereby suppressing prey-abundance and increasing their body size. Dixon (2000) opined that variation in body-size within the species and gender might be associated with the relative effects of food quality and quantity. Furthermore, smaller-sized ladybirds may exploit the aphid colonies earlier, which may later be overtaken by the large ladybirds when aphid densities increase (Dixon 2007). Sloggett (2008) argued that ladybirds' body size might not be just a function of aphid density, but other complex interactions between density and prey size are also operational. This further raises the question of whether con-

tinuous exposure of aphidophagous ladybirds to aphid abundance may increase the growth rate and have evolutionary significance. Most species with high biocontrol potential are large and highly fecund, which are favoured by natural selection, particularly in food-abundant habitats (Brown and Sibly 2006). Large species have a reproductive advantage over smaller indigenous species in prey-rich habitats (Kajita and Evans 2010).

Coccinella transversalis Fabricius is a predator (Coleoptera: Coccinellidae) of many insects and acarine pests, particularly, aphids (Omkar and James 2004; Omkar and Pervez 2004b; Maurice et al. 2011). Manipulation of its reproductive parameters may promote its abundance (Michaud et al. 2013). It coexists with other coccinellids and mostly dominates the aphid predatory guild (Omkar et al. 2005a, b) and together with coccinellid, *Propylea dissecta* (Mulsant) may synergistically suppress populations of *Aphis gossypii* (Glover) (Omkar and Pervez 2011). We found adults and larvae of *C. transversalis* preying on rusty plum aphid, *Hysteroneura setariae* (Thomas) infesting creeping bluegrass, *Bothriochloa insculpta* (Hochst.). This aphid is a cereal pest, attacking rice, wheat, sugar cane, maize and soya bean crops on the Indian sub-continent (Kale et al. 2020). In a banker plant system, *H. setariae* reared on grasses, can be used as a non-pest prey to build-up ladybird populations (Rattanpun 2017). Hence, we designed a laboratory experiment to determine (i) the searching efficiency of adult male and female *C. transversalis* feeding on *H. setariae* (ii) killing power of adult ladybirds associated with their aggregation, and (iii) the influence of the intraspecific competition for food on the adult bodyweight and its implications.

Materials and Methods

Insect culture and maintenance

We sampled and collected adults of *C. transversalis* from *H. setariae* infested fields of *B. insculpta* near our college campus, Kashipur, India (30.2937°N, 79.5603°E). We brought them to the laboratory and paired adult male and female ladybirds in Petri dishes (9.0 cm diameter × 2.0 cm height) containing an *ad libitum* amount of *H. setariae* infesting host plant twigs. The females mated and laid eggs in clusters that were isolated and kept in other Petri dishes (size as above). We transferred these Petri dishes to an Environmental Test Chamber (ETC) (*REMI, Remi Instruments*), maintained at 25 ± 1 °C, 65 ± 5 % R.H and 12L : 12D. The eggs hatched and the first instar larvae were placed in 500 ml *Borosil* glass beakers containing sufficient supply of aphid infested twigs. Five first-instar larvae were kept in each beaker and reared on aphids until adult emergence. We replenished the aphids daily to avoid contamination. The newly eclosed F_1 adults were sexed and isolated in separate Petri dishes, (size as above) for use in the experiments.

Experimental design

Fifteen-day-old adult male *C. transversalis* was taken from the stock and starved for 12 hours to standardize its hunger. Thereafter, we weighed it (W_1) using an electronic balance (SHIMADZU, Model ATX-224, 0.1 mg precision) and kept it in a 500ml glass beaker containing 200 third-instar nymphs of *H. setariae* (as prey). A piece of folded moist filter paper was also kept in the beaker to provide moisture. We covered the beaker with fine muslin cloth fastened by a rubber band. We transferred this beaker to ETC maintained at the abiotic conditions mentioned above. After 3 hours of exposure, we removed the beaker from ETC and counted the number of live aphids to determine the number of aphids consumed (N_a). The ladybird was weighed again (W_2) (as above) to estimate the gain in weight ($W_e = W_2 - W_1$, *i.e.* final weight of adult – the initial weight of adult). This experiment was replicated ten times ($n = 10$). We repeated the experiment at predator densities of 2, 4, 8, and 10. Thereafter, the entire experiment was repeated using the above predator densities of 15-day-old adult female *C. transversalis*. The data were subjected to the following data analysis.

Data analysis

Nicholson–Bailey model gave the following equations (1) and (2):

$$N(t+1) = \lambda N(t) \exp[-aP(t)] \quad (1)$$

$$P(t+1) = cN(t)[1 - \exp(-aP(t))] \quad (2)$$

where $N(t)$ is the number of hosts (prey) at time t , $P(t)$, the number of predators at time t , λ is the host reproductive rate, and a is the area of discovery. To estimate the area of discovery, the above model (2) can be rearranged (Hassel 1978) after assuming that $c = 1$, as:

$$a = \frac{1}{P} \log e \frac{N}{(N - Na)} \quad (3)$$

where a is the area of discovery, N is the initial aphid density, Na is the number of aphids consumed, and P is the predator density. We used the above-rearranged model (3) to relate the area of discovery to prey density. After estimating the area of discovery, Hassell and Varley (1969) model (equation 4) was used to estimate Quest constant (Q), while mutual interference (m) constant was determined from the slope of regression of $\log a$ (area of discovery) on $\log P$ (predator density).

$$a = QP^{-m} \quad (4)$$

Equation (4) can be linearized by using logarithms as follows:

$$\log a = \log Q - m \log P \quad (5)$$

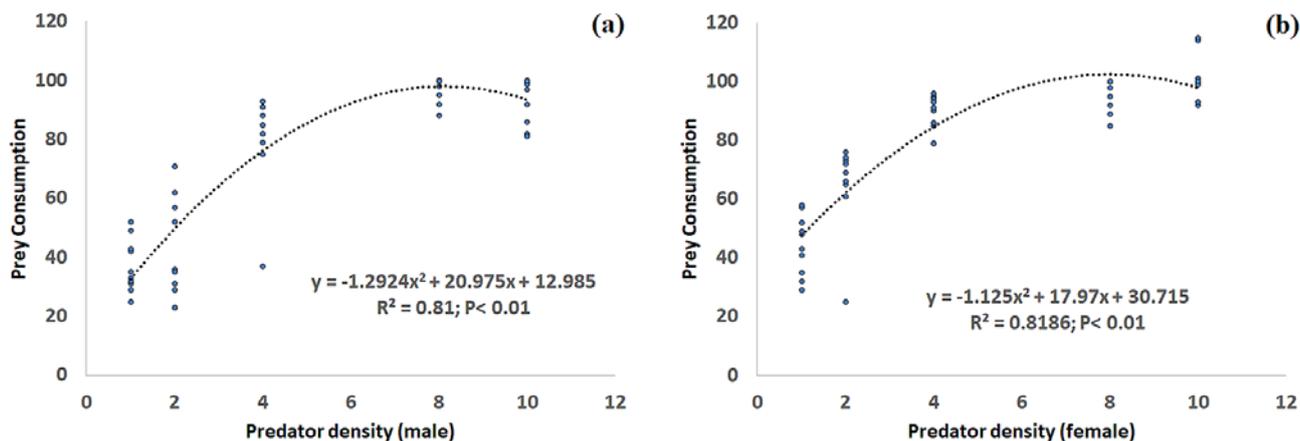


Fig. 1 Relationship between prey consumption and predator density for (a) male and (b) female *C. transversalis* fed the aphid, *H. setariae*.

k -value, which is the measure of the ‘killing power’ (Ooi 1980) was also estimated by taking the difference between the logarithms of aphid population before and after prey consumption (Varley et al. 1973) at various predator densities using equation (6).

$$k\text{-value} = \log_{10} (N / S) \quad (6)$$

The number of prey consumed per predator by adult male and female ladybirds at different predator densities was subjected to one-way ANOVA using statistical software SAS 9.0 (SAS 2002). The means were compared using Tukey’s test of significance. We also subjected the prey consumed per predator at particular predator densities for both adult male and female ladybirds to a two-sample t-test using SAS 9.0. All data were tested for normality and variances using the Shapiro-Wilk test. The (i) prey consumption, (ii) area of discovery, (iii) killing power and (iv) mean weight gained or weight gained per predator with the increase in predator density were further subjected to regression analysis to discover the relationship between these variables using SAS 9.0. The log area of discovery and the log predator density

were subjected to linear regression in order to determine the mutual interference and Quest constants using SAS 9.0.

Results

The prey consumption of the adult male and female *C. transversalis* increased curvilinearly with increase in predator density (Fig. 1). The female ladybirds consumed a significantly greater number of aphids than the males ($t = 3.95$; $P < 0.01$; d. f. = 94). The prey consumption per predator decreased significantly with increase in predator density (Table 1). The difference in the prey consumption of males and females was only significant when the number of ladybirds was two ($t = -3.11$; $P < 0.01$; d. f. = 17) and ten ($t = -2.27$; $P < 0.05$; d. f. = 17) (Table 1). The area of discovery of male and female beetles decreased with increase in predator density (Fig. 2). The log values of area of discovery of male ($r^2 = 0.5703$; $P < 0.05$) and female ($r^2 = 0.8099$; $P < 0.01$) beetles showed a significant linear relationship with increase in log predator density ($P < 0.01$). The mutual interference constants for adult

Table 1 Prey consumption per adult male and female *C. transversalis* at various predator densities.

Predator density	Adult Female	Adult male	t-value	P-value	d. f.
1	44.40 ± 10.16 a	37.10 ± 8.93 a	-1.71	0.160	17
2	32.10 ± 7.37 b	20.95 ± 8.60 b	-3.11	< 0.010	17
4	22.20 ± 1.57 c	20.08 ± 4.07 b	-1.58	0.148	17
8	11.90 ± 0.69 d	12.16 ± 0.53 c	0.64	0.533	16
10	10.14 ± 0.76 d	9.36 ± 0.78 c	-2.27	< 0.05	17
F-value	63.74	34.15			
P-value	P < 0.0001	P < 0.0001			
d. f.	4, 49	4, 49			

Data are Mean ± S.D.; Tukey’s range = 4.02

Means compared by using different letters in rows or columns to denote statistically significant differences.

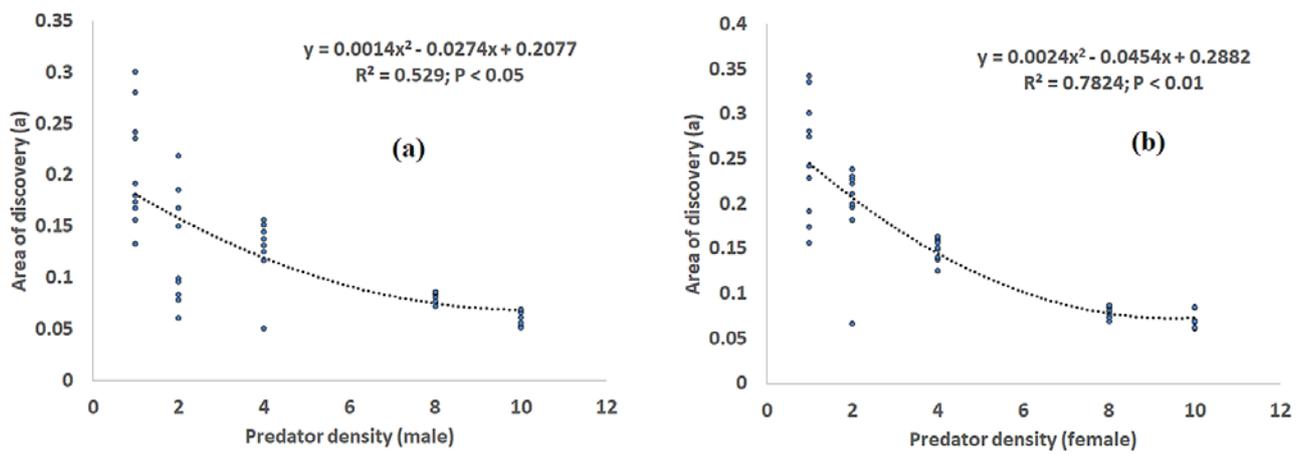


Fig. 2 Relationship between area of discovery and predator density for (a) male and (b) female *C. transversalis* fed the aphid, *H. setariae*.

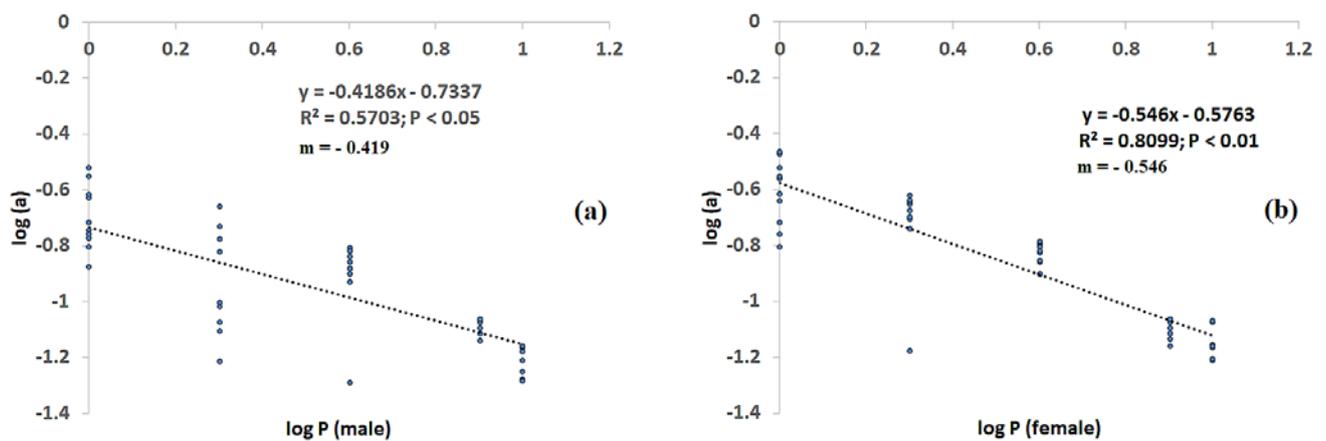


Fig. 3 Mutual interference (*m*) derived from the relationship between logarithm of predator density and area of discovery for the (a) adult male and (b) female ladybird, *C. transversalis* fed the aphid, *H. setariae*.

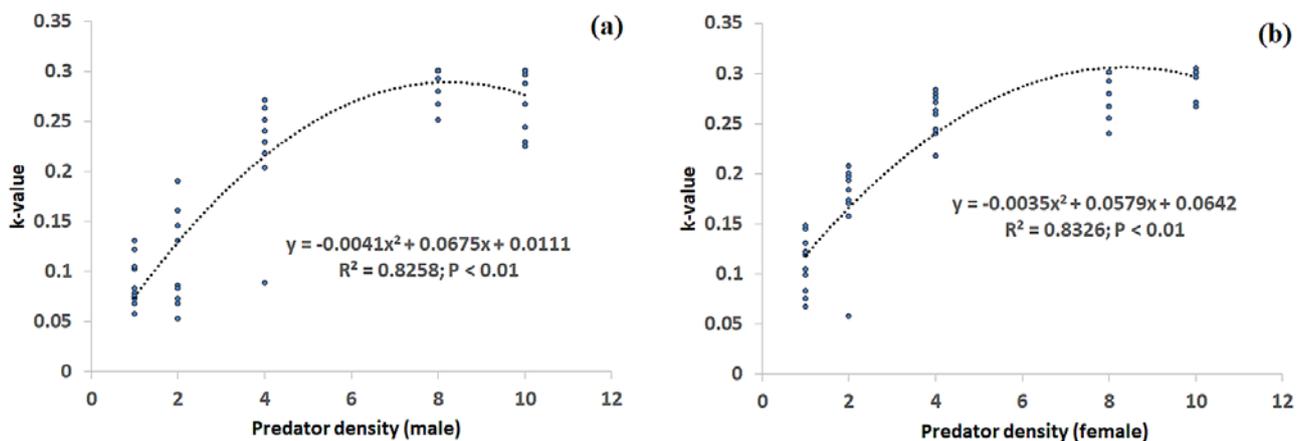


Fig. 4 Relationship between killing power (*k*-value) and predator density for (a) male and (b) female *C. transversalis* fed the aphid, *H. setariae*.

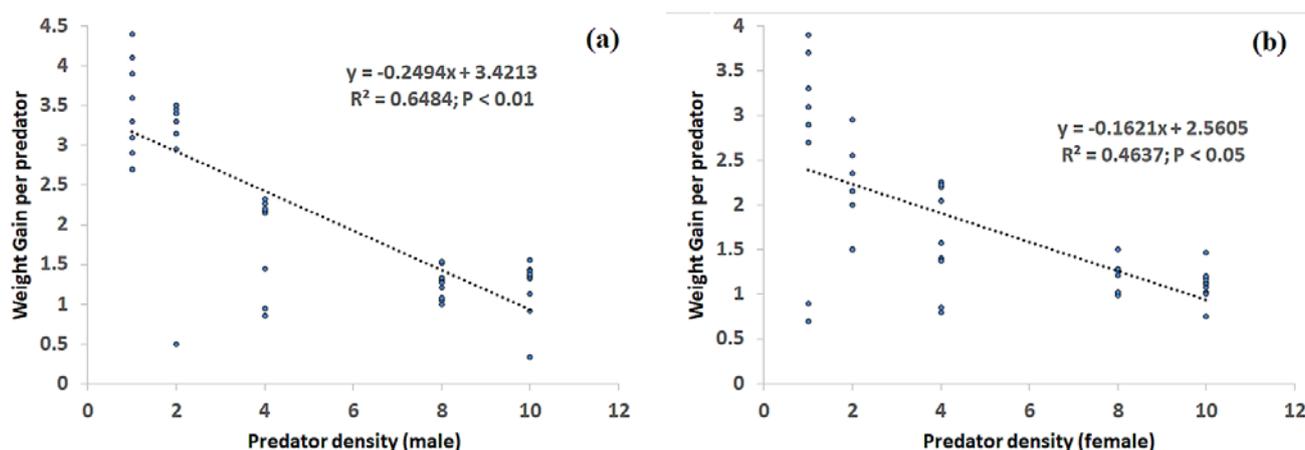
male and female ladybirds were -0.419 and -0.546 , respectively (Fig. 3), while the quest constants were 0.21 and 0.25 , respectively. The killing power of the ladybird denoted by the *k*-value, increased curvilinearly with increase in predator density (Fig. 4). The mean initial

weight (W_1) and the mean final weight (W_2) differed significantly both in the case of adult males and females of *C. transversalis* (Table 2). The weight gained per predator decreased linearly with increase in predator density of both male and female ladybirds (Fig. 5).

Table 2 Mean initial weight (W_1) and Mean Final weight (W_2) of adult male and female *C. transversalis* at different predator densities when provided with a constant number (200) of the aphid, *H. setariae* as prey.

Predator density	Adult male			Adult female		
	Mean Initial weight (W_1)	Mean Final weight (W_2)	t-value and probability	Mean Initial weight (W_1)	Mean Final weight (W_2)	t-value and probability
1	13.89 ± 1.58	16.61 ± 2.06	t = -3.32; P < 0.01; d.f. = 16	19.08 ± 2.74	22.54 ± 3.12	t = -2.64; P < 0.05; d.f. = 17
2	13.93 ± 0.99	15.99 ± 1.26	t = -4.09; P < 0.001; d.f. = 17	20.31 ± 0.69	23.56 ± 1.08	t = -7.52; P < 0.0001; d.f. = 15
4	14.36 ± 0.51	16.05 ± 0.75	t = -5.91; P < 0.001; d.f. = 15	21.74 ± 1.35	23.62 ± 0.91	t = -3.66; P < 0.01; d.f. = 15
8	14.56 ± 0.62	15.23 ± 0.67	t = -4.02; P < 0.001; d.f. = 17	20.33 ± 0.29	21.59 ± 0.34	t = -9.00; P < 0.0001; d.f. = 17
10	14.62 ± 0.64	15.73 ± 0.70	t = -4.09; P < 0.001; d.f. = 17	21.71 ± 0.61	23.03 ± 0.65	t = -4.72; P < 0.001; d.f. = 17

Data are Mean ± S.D.

**Fig. 5** Relationship between weight-gain per predator of adult (a) male and (b) female *C. transversalis* subject to different levels of competition for the aphid, *H. setariae*.

Discussion

The consumption of prey by adult males and females of *C. transversalis* increased with predator density, indicating that in aggregating they increase prey mortality. However, the rate of increase per predator declined with increase in the number of predators due to mutual interference negatively affecting prey consumption, as reported in previous studies (Bayoumy and Michaud 2012; Bayoumy et al. 2014). The females usually consumed more aphids than the males, which might be attributed to their larger body-size and energy demands for egg production (Lucas et al. 1997; Šipoš et al. 2012). Females of aphidophagous ladybirds need more energy to forage for aphids, search for ovipositional sites and lay eggs, while males just need energy to maintain themselves and to search for females (Hemptinne et al. 1996). Such females tend to search more actively when aphids are scarce or absent than when they are abundant (Evans and Dixon 1986). Hence, female ladybirds locate and consume more aphids than males. In addition, female foraging and prey-consumption may be directly linked to the numerical response, *i.e.* lay as many eggs as possible, unlike

the males whose activities are seemingly dominated by searching for and copulating with females (Evans 2003). Ives (1981) note that the residence time (*i.e.* time spent in an aphid colony) of the female coccinellids, *Coccinella septempunctata* L. and *Hippodamia variegata* (Goeze), is greater than that of males, and aggregation of females was positively correlated with aphid density.

The area of discovery of foraging adults decreased with increase in their density indicating a decline in searching efficiency, the effect of which was greater at high predator-densities implying greater interference between predators. This further indicates that aggregation in a prey patch may result in an increase in mutual interactions that may reduce their effect on prey mortality. Hassell (1971) suggests that each predator might spend less time searching for prey and more time interacting with conspecifics when predators aggregate in patches of prey. We confined the ladybirds in an experimental arena, which resulted in a high incidence of mutual interactions. This indicates that the results may not be relevant to what happens in the field, however, as in patches with a low density of aphids ladybirds may experience a lower incidence of mutual interference with similar outcomes.

Hence, when there are few conspecific ladybirds present in a patch mutual interference will be low and prey mortality higher and *vice versa*. Thus, it is important to avoid releasing high numbers of conspecific ladybirds, which might result in high levels of mutual interference and have a negative effect on aphid suppression, decrease in mean weight gain and cannibalism of larvae and pupae. Hence, optimal foraging and the laying eggs (Kindlmann and Dixon 1993) may not occur when ladybirds are abundant, but when scarce it is advantageous in terms of gain in bodyweight and avoidance of cannibalism.

The area of discovery and mutual interference are indicative parameters of the total time spent interacting with other conspecific predators (Rogers and Hassell 1974). Siddiqui et al. (2015) report that mutual interference of slow and fast developing ladybird, *Propylea dissecta* (Mulsant) were -0.394 and -0.808 , respectively, indicating that fast developers search more efficiently and spend less time interacting with conspecifics. Fast developers tend to eat faster than slow developers and are heavier and lay more eggs than the latter (Singh et al. 2014; Dixon et al. 2016). Mutual interference values for unparasitized and parasitized larvae of *Nephus includens* (Kirsch) were -0.44 and -0.92 respectively, indicating that interference reduces the foraging capacity of parasitized more than that of unparasitized larvae (Bayoumy and Michaud 2012). Similarly, the mutual interference values for adult male and female *C. transversalis* were -0.43 and -0.72 , respectively, which indicates that females are better foragers and interfere lesser than males.

We recorded a curvilinear increase in aphid consumption with increase in predator density. Bayoumy et al. (2014) note that the killing power of the acarophagous ladybird, *Stethorus gilvifrons* Mulsant increases with predator aggregation. Adult females consume more aphids than males because they are bigger than males. The mean initial and final adult weights of *C. transversalis* differed significantly indicating that prey consumption is a function of bodyweight. Ladybird abundance in an aphid-resource rich environment may result in an increase in adult body-size. Individual ladybirds vary in body-size for dietary and genetic reasons. It is widely held that body-size is positively correlated with fitness and is driven by diet (Stearns 1992). Hence, it is likely that the heaviest adults will have a selective advantage. However, small generalist ladybirds, which feed on a wide range of species of aphids, may have an advantage when aphids are scarce (Sloggett 2008). We also recorded that regardless of gender, predator abundance tends to be associated linearly with decrease in the weight gained per predator, which was significantly greater when the number of predators was low, which might indicate that mutual interference was lower and prey consumption per predator higher than when number of predators was high. Hence, selection should favour adults, which as described by Dixon (2000) are able to avoid laying eggs in patches of aphids already being exploited by ladybirds as

it not only results in an increase fitness but also a reduction in mutual interference between the larvae. However, further research is needed to address this issue.

It is concluded that (i) the searching efficiency of *C. transversalis* decreased with increase in predator density, (ii) mutual interference negatively affected prey consumption especially that of adult males, (iii) the difference in the aphid consumption of females and males became more skewed in favour of females with increase in predator density, and (iv) the gain in bodyweight per predator decreased with increase in the number of ladybirds.

Acknowledgements

Authors thank Prof. A.F.G. Dixon, Emeritus Professor, School of Biological Sciences, University of East Anglia, Norwich, UK and Prof. Pavel Kindlmann, Charles University, Prague, Czech Republic for improving the English and providing fruitful comments and suggestions, Dr. A. Betsy for improving the draft at the initial stage, and Science and Engineering Research Board, Department of Science and Technology, New Delhi for funding this research (EMR/2016/006296).

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THE STATUS OF COMMONS IN THE CHANGING LANDSCAPE IN THE CZECH REPUBLIC

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ABSTRACT

Commons were ancient pastures, which once occurred in every village in many countries, including the Czech Republic. They have been a landscape and social phenomenon for decades. However, social and economic changes brought an end to community ownership and traditional management of these commons. The number of commons has been decreasing since the middle of the 19th century and currently very few remain. This paper evaluates the status of former commons in 35 cadastres in south-western Bohemia and describes the changes they have undergone in the last two hundred years. Three historical periods were identified as the main drivers in the changes in the status of commons. We started with a period from the middle of the 19th century to the 1950s, the second from 1950s to 1990s and the last from 1990s to 2019. Aerial images and field surveys revealed that 93% of former commons disappeared due to afforestation, conversion to fields and natural succession occurring on abandoned commons. The social and economic aspects associated with these changes are mentioned. Some of the commons are part of the Territorial system of landscape ecological stability (Ecological networks) and we suggest that more of the remaining commons should be included in this network. They could play a role in maintaining biodiversity and providing stepping stones in a uniform agriculture landscape. We propose to evaluate the conservation and ecosystem value of these commons in more detail and set up the appropriate management essential for the preservation or restoration of commons, an indisputable part of our biological and cultural heritage.

Keywords: aerial images; commons; historical maps; land use changes

Introduction

“The tragedy of Commons” by Hardin (1968) inspired this study, however we see the tragedy of commons from a different point of view. We tried to determine whether the current state of the commons can be described only as a tragedy or whether there is hope that commons provide opportunities for improving uniform landscapes. This study evaluates the status of commons over nearly two hundred years.

The Central European region has been significantly affected by human activities for centuries. Wildness was gradually transformed into a cultural landscape permanently managed by humans. The richness and diversity of rural landscapes is a European phenomenon and a consequence of the long history of the Old Continent landscape. However, recently the rural landscape in Central Europe changed significantly. The scale of change has increased and accelerated during the last decades. Transformation of agriculture, new technologies and socio-economic changes are the main drivers of these changes in land use (Mander et al. 2004). Grazing land is one of the most affected habitats (Palang et al. 2006). Grazing animals, recognised as important drivers of Central European landscape structure and regional diversity, have almost completely disappeared in recent decades.

Since the Middle Ages, common pastures, often called commons, used to be a common feature of the Central European landscape. They are a traditional phenomenon in many aspects, including biological and cultural. Commons were nutrient poor, waterlogged or stony localities,

not suitable for agriculture, and were usually used daily for mixed grazing. The daily regime was controlled by a municipal shepherd, who brought the herd to the common and back to the stables every day. Long term low-intensity mix grazing resulted in commons being localities rich in different habitats and species, and home to many protected species. Thanks to small fertilizer input and extensive grassland use, common pastures are semi-natural grasslands with a high conservation value, which are often called “biodiversity hotspots” or “biodiversity refugia” (e.g. Rook and Tallowin 2003 or Hodgson et al. 2011). The importance of commons for the Central European fauna and flora has already been confirmed by several studies. For example, the importance of commons as bird refugia is confirmed by Schwarz et al. (2018). Berg et al. (2011) emphasize the importance of commons for the conservation of large butterfly populations. Their high biological value is enhanced by their high conservation value in this area. Many small protected areas (i.e. nature reserves, nature monuments) were established in previous commons.

This study evaluates the status of former commons in south-western Bohemia. The area of interest includes the wider surroundings of the village Těchonice, where many commons were preserved or restored thanks to the enthusiasm and care of local residents. The commons called “Těchonické draha” are the Arch of biodiversity hosting many specific habitats and species. To better understand the status of commons and how they have changed over time, we analysed the status of commons in four periods: the 1850s, 1950s, late 1990s and 2019 and discuss the

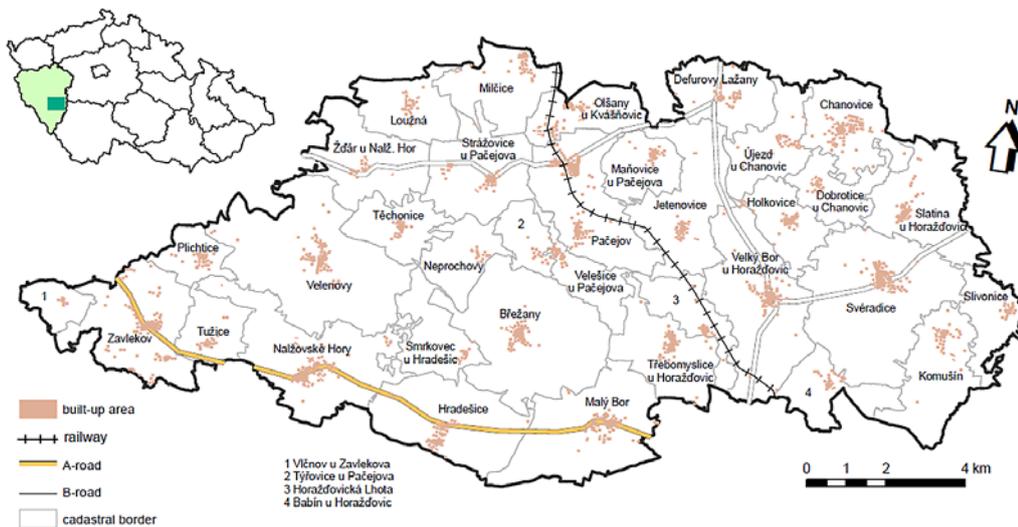


Fig. 1 Area studied.

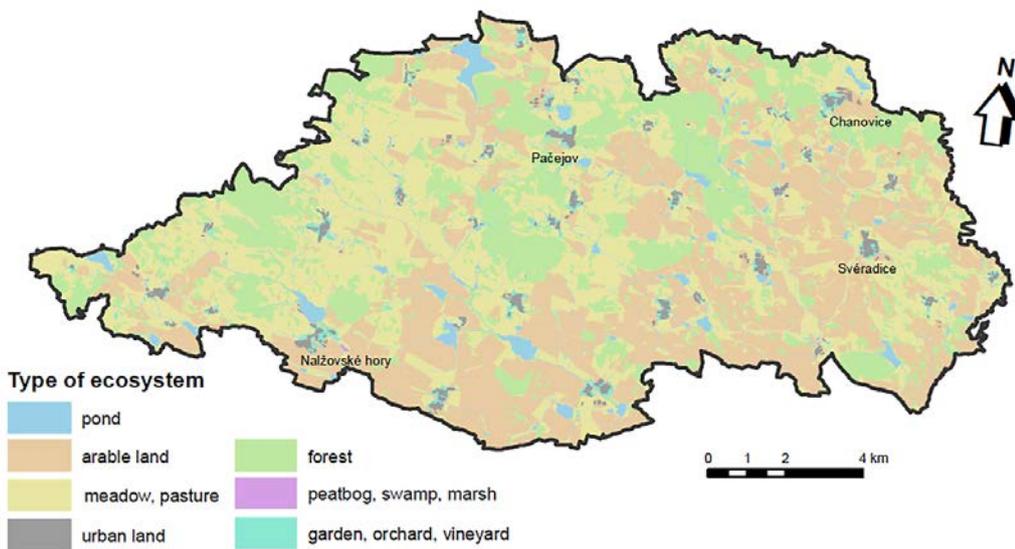


Fig. 2 Map showing the distribution of particular habitats in the area studied.

changes that occurred in each period. Finally, we discuss the potential of commons for improving the quality of the current landscape and mitigation of effects of climate change.

Methods

Study area

The area studied is located in the Pilsen region, in the south-western part of the Czech Republic (Fig. 1). It has an area of 17 km² and consists of 35 cadastral areas. The largest settlements are Nalžovské Hory with over 1000 inhabitants and Chanovice and Pačejov with over 700 permanent inhabitants. The area is characterized by a rural landscape with ponds, many pastures, small villages and low density of transport infrastructure. The only trans-

port infrastructure through the area is the railway corridor Pilsen – České Budějovice. The area selected is quite similar to other parts of the Czech Republic (e.g. some regions at low altitudes in the Šumava Protected Landscape Area or the Vysočina region (Culek et al. 2013)).

Climatic conditions in most of the area studied is mildly warm and warm in the southern part (Cenia 2017). Most of the area is composed of intrusions of Central Bohemian pluton, especially granodiorites, or granites, which often rise up in the terrain in the form of large boulders or rocks. The soil cover consists mainly of acidic cambisols. Forests, mostly of spruce or pine, cover about 20% of the area (Culek et al. 2013). In general, fields are present in a non-forest landscape, in which pastures and meadows are less abundant. However, meadows and pastures predominate in the north-western part of this area (Fig. 2).

Processing of data

This study involved: 1) the vectorization of data, 2) comparison of aerial images with other maps, 3) field research in autumn 2019 and spring 2020.

Because the data needed is not yet available for the area studied, we first created a vector layer of former municipal pastures. Imprints of historical maps of Stable Cadastre for half of the 19th century (Semotánová 1998) provided by ČÚZK (2020) were used. Four categories of commons based on their size were distinguished: (i) micro – with a size of 0.5 ha, (ii) small – 0.5–1.5 ha, (iii) medium – 1.5–5 ha and (iv) macro – more than 5 ha. In order to analyse their status, the layer of segments of commons was compared with aerial images from 1951, 1999 and 2019. Aerial photographs from 1951 indicate the traditional structure of the landscape before collectivization and the creation of agricultural cooperatives, by which the communists fundamentally changed the economy in the countryside. Aerial photographs from 1999 are of the landscape at the end of the first decade of post-communism, when agricultural land was returned to the former owners, sold or privatised. Finally, the current situation is recorded in the aerial photographs from 2019.

The following commons were categorized in each period:

- 1) preserved commons – more than 2/3 of which are covered with a mosaic of grassland vegetation;
- 2) abandoned – more than 2/3 of which are covered with naturally regenerated trees or shrubs as they are no longer used for grazing animals;
- 3) afforested – more than 2/3 of which were afforested;
- 4) converted to fields – more than 2/3 of which were improved by melioration, drainage or other technical adjustments and transformed into arable land;
- 5) built up – more than 2/3 of which was covered with houses or other infrastructures (e.g. agriculture buildings, playgrounds, municipal waste landfills etc.);
- 6) other – more than 2/3 of which was converted to something other than that listed above.

The database included all the above data and used in the following research.

Then, the commons in current aerial photographs categorised as preserved and larger than 0.5 ha (i.e. size category (ii) small, (iii) medium and (iv) large), were selected. A layer consisting of these commons was overlaid with the following maps:

- consolidated layer of ecosystems,
- Natura 2000 habitats,
- protected species listed in the Nature Conservation Finding Database.

Finally, the status of preselected commons was verified in the field. Field surveys were carried out in autumn 2019 and spring 2020 to determine whether the characteristics based on the aerial photographs (i.e. the size of the open area and the assumed mosaic nature of the habitat) correspond with that observed in the field. The field survey confirmed or refuted the inclusion of a common

on the list of preserved commons. This verification also helped us to determine whether the aerial photographs could also be used to identify preserved commons.

Results

Commons in the middle of the 19th century

The typical rural landscape in the middle of the 19th century consisted mainly of small private fields, sporadically distributed in extensive forests, along with commons and generally little urbanisation. For centuries, the acreage of arable land increased at the expense of forests. The middle of the 19th century is when the area of forest in our landscape reached the historically lowest value and there were no further possibilities for increasing the area of agricultural land and the agricultural landscape was formed (Bičík 2010 in Vachuda 2017).

In the middle of the 19th century, large commons occurred further from the centres of villages than the small commons that occurred irregularly along paths, between fields, around houses and in gardens. In many areas, all these typical formations were evenly represented in the rural landscape.

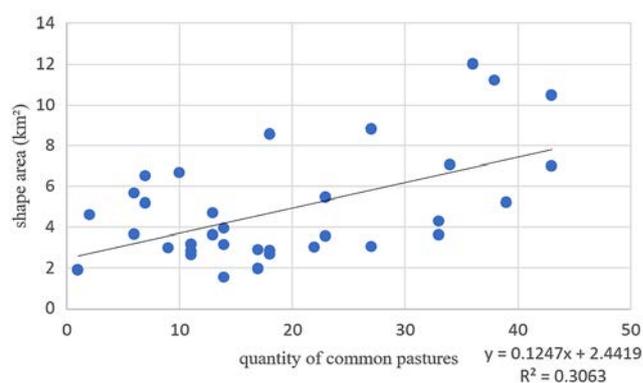


Fig. 3 The relationship between the number of commons and the size of the cadastral area.

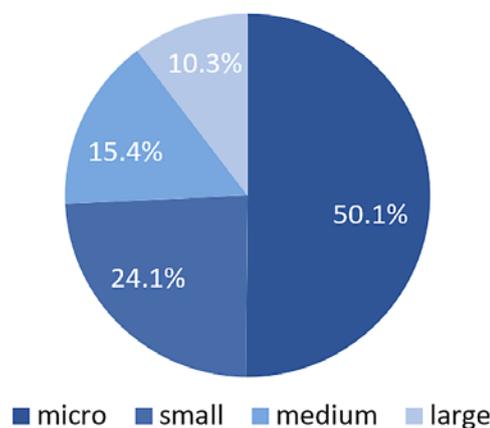


Fig. 4 Pie diagram illustrating the percentage of commons in each of the size categorises.

Based on historical maps from the middle of the 19th century, 668 commons were in the area studied. As expected, the number increased with the size of the cadastral area, but the relationship is not very strong (Fig. 3).

The size of commons varies markedly and they are not distributed uniformly in terms of size categories (Fig. 4). More than 50% of all commons are smaller than 0.5 ha. These small and often narrow commons were usually used as corridors for moving grazing animals from one pasture to another. Herds of cattle used them for a short stop during their regular trips to large pastures and therefore they occur more frequently than large commons.

The early 1950s

The aerial photographs from the early 1950s reflect the situation in Czechoslovakia after World War II and at the beginning of the socialist era. Land reforms started after the Communist revolution in 1948. However, aerial photographs from the early 1950s reveal that the area studied has not changed significantly in that individual plots, small fields, forests and other types of individual properties were still present.

Compared with the situation 100 year ago, the percentage of the land classed as agricultural is similar and only structural changes occurred in the 1950s. During this 100-year period, the area and number of commons decreased only slightly. More than 2/3 of the former commons were preserved (Table 1) and they made up an important part of the landscape. Borders of the commons were usually clear and rarely violated. The biggest percentage of commons was lost to afforestation, which occurred at the beginning of the 19th century (Bičík 2010 in Vachuda 2017). About 5% of former commons was converted into fields or meadows. Occasionally, some drainage or landscaping (e.g. removing big stones or levelling of the surface) were necessary. However, the extent of this landscaping was small compared with what happened in the coming decades. The area of agricultural land increased, but not very significantly. Some commons were built on and others abandoned and overgrown in the course of natural succession. Based on the aerial photographs, the preserved commons were those

that were not afforested, built on or abandoned and then subject to natural succession.

The late 1990s

The status of commons in the last years of the 20th century is the result of four decades of socialist agriculture and not always appropriate management of the Czech landscape. However, at the end of the 1990s, the effects of economic and property changes, which were implemented after the Velvet Revolution in 1989 (including privatization, abolition of agricultural cooperatives, reduction in arable land, etc.), are also evident. Until the 1990s, the agricultural policy in Czechoslovakia was influenced by farm nationalization and that resulted in significant changes in the landscape, predominantly in the percentage of arable land. In the early 1990s, landscape was affected by the change in ownership, both with restitution and privatisation. Especially agricultural land, which was divided among a large number of owners, but only a fraction of them farmed their land again (Kabrda and Bičík 2010). That led to abandonment, renting and changes in the use of these lands. When the restitution and restructuring was complete, comprehensive land adjustments began. In the aerial images, changes in the use of commons are very noticeable at this time.

The decrease in the number and acreage of commons continued. More than half of all commons disappeared and only one third were preserved (Table 1). The trend in transforming commons into agricultural fields escalated. In the late 1990s, a quarter of former commons were already converted into fields and being used as an agriculture field or a part of a large agricultural complex. Another 22.3% of commons was abandoned and left to natural succession. From the middle of the 20th century, there was a very rapid increase in population resulting in a 6% increase in built-up areas on unused parts of former commons. In addition, the percentage of afforested commons increased from 8.1% in 1950s to 12.1% in the late 1990s (Table 1).

The status of commons in 2019

During the first 20 years of the new millennium there were still significant changes, which resulted in the transformation of former commons into other functional segments of landscape. The changes were not as significant as in the previous period. However, we must consider the length of the period, which was only two decades. The main driver of the transformation of commons in this period was the increase in the number of abandoned commons. In 2019, more than one third of commons had vanished due to natural succession. Abandoned, unmanaged commons became overgrown naturally because of a sequel of privatization in the 1990s and unclear ownership or speculation over the sale of the land. In addition, because in previous times the commons were often rocky or waterlogged localities with inaccessible terrain, it proved difficult to find an alternative use for them.

Table 1 Percentage of preserved commons present at different times from 1850s to 2019.

State of commons	1850s	early 1950s	late 1990s	2019
preserved	100%	76.5%	30.2%	12.1%
built on	0%	2.5%	6.0%	6.3%
converted to fields	0%	5.1%	25.1%	29.0%
afforested	0%	8.1%	12.1%	12.7%
abandoned	0%	3.7%	22.3%	34.7%
other	0%	0.1%	0.9%	1.3%
combination	0%	3.9%	3.3%	3.7%

Table 2 Changes in percentage of different land covers recorded in the different periods.

Status of commons	1850s	early 1950s	late 1990s	2019
preserved	100%	-23.5%	-46.3%	-18.1%
built up	0%	2.5%	3.4%	0.3%
converted to fields	0%	5.1%	20.1%	3.9%
afforested	0%	8.1%	4.0%	0.6%
abandoned	0%	3.7%	18.6%	12.4%
other	0%	0.1%	0.7%	0.4%
combination	0%	3.9%	-0.6%	0.4%

Large agricultural complexes, commercial forests and urban areas were the most abundant structural elements in the landscape in 2019. The borders of former commons are not clearly visible and if so, mostly it is a border of an overgrown area, where natural succession has been occurring for a long time. Currently only about 12% of former commons remain (Table 1), but they have a high conservation value because they host a mosaic of vegetation.

Almost two centuries of change

There have been significant changes in the status of commons in the area studied since the middle of the 19th century. This period was divided into three, in which the changes are visible and can be easily evaluated. An illustrative series of pictures showing the transformation of commons over almost two centuries is in Appendix 1.

Decrease in the number and acreage of commons during these three periods was not uniform (Table 2). During the first period (1850s–1950s), the largest percentage of commons was lost due to afforestation. We assume that this was due to the beginning of the large

afforestation at the end of the 19th century. In the following periods, this transformation was never that visible.

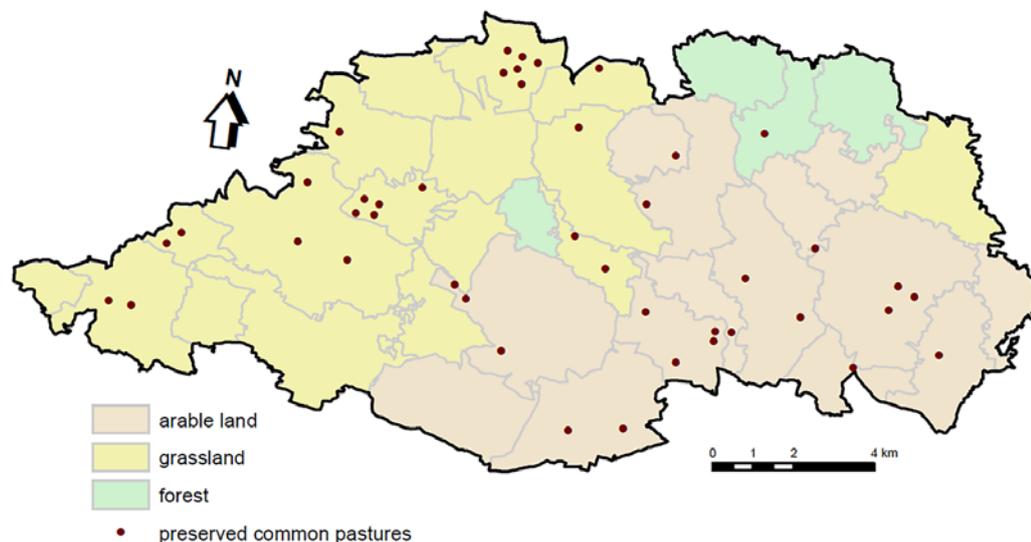
The biggest changes occurred during the second period (1950s–1990s), although this period was significantly shorter than the first. The main driver of change was amelioration, when almost a fifth of the commons was drained, big stones removed, ploughed and converted into agriculture fields and meadows (Table 2). Almost the same percentage was abandoned and left to natural succession.

In the last period, the above factors were of little significance except for abandonment (Table 2). Inappropriate management in previous times and ownership changes in the 1990s resulted in an increase in them being left to natural selection.

Preserved commons

Finally, analyses of maps together with the two-step field verification (autumn and spring) helped us to identify 49 preserved commons in the area studied. They were open with a mosaic of vegetation and their borders were well preserved and visible. Their distribution in the area of interest is very irregular and the predominant ecosystem in the cadastral area has no effect on the number of preserved commons (Fig. 5).

Our field surveys confirmed that many of the preserved commons consist of a mosaic of significantly valuable habitats, which are occupied by rare and protected species. The existence of a mosaic of habitats, mostly in extensively managed commons, in the current monotonous landscape is highly valuable from the conservation point of view. It is well established that these commons contribute to the conservation of species and habitat biodiversity. Their presence in the landscape is therefore crucial. The majority of the preserved commons are part of the Terrestrial system of ecological stability (Hrnčia-

**Fig. 5** Map showing the predominant ecosystem in each cadastral area and the distribution of preserved commons.

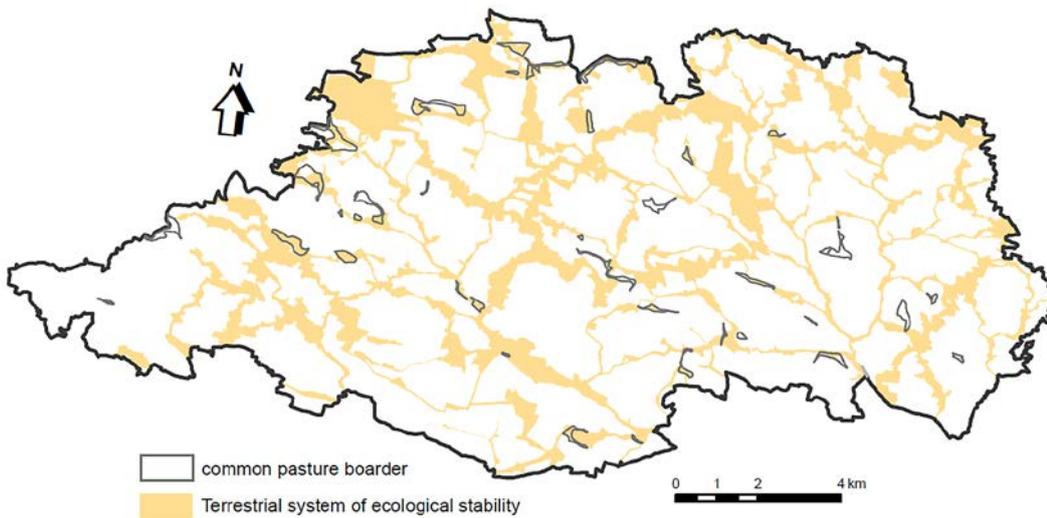


Fig. 6 Map showing the location of preserved commons and the Terrestrial system of landscape ecological stability.

rová et al. 2009; ecological networks in the sense of the Czech Nature Conservation Act No. 114/1992; Fig. 6). In total, 51% of the preserved commons mapped are local bio centres or local bio corridors.

Discussion and Conclusions

This study evaluated the current status of former commons, which were identified based on the maps of the Stable Cadastre, unique maps that cover the entire area of Bohemia, Moravia and Silesia (Semotánová 1998). Using these maps, it is possible to reconstruct the landscape present in the middle of the 19th century with a high degree of accuracy. They are an important source of knowledge on the character of the historic landscape and their accuracy enables one to digitally process and implement them in the geographic information system (GIS), which opens up further possibilities for analysing the structure of the historic landscape and comparing it with the current state (Brůna et al. 2004, 2006). Nedbal et al. (2008) and Brůna et al. (2010) state that the data from the Stable Cadastre are most suitable in terms of precision for monitoring the state and changes in the non-forest landscape. It was an essential basis for determining land taxes, and therefore both mapmakers and landowners were very interested in the exact details of the land and the determination of culture, i.e. the current use of land. The suitability of the maps of the Stable Cadastre for monitoring landscape changes in areas of increased conservation interest was also confirmed by this study, which focused on a selected type of land – commons.

In the area studied, which included 35 cadastral units, 668 former commons were identified. This area was selected because of the many well-preserved commons around Těchonice village, which is located in the mid-

dle of the area studied. This rural countryside with poor infrastructure and many commons was a suitable study area. However, landscape structure in this area is similar to other Czech regions and the methodology used can be easily implemented in other similar areas. Worth noticing is that only a small number of commons larger than 1.5 hectares were recorded in this study. These commons can be grazed by large and, above all, more diversified herds, which results in a specific type of farming beneficial from the point of view of maintaining diversity. In addition, to the information on the size and frequency of commons in the land register, it was possible to determine whether and how the size of the land area and the size of the commons are related. We found only a slightly positive correlation, which was certainly influenced by the fact that individual cadastres not only differ in area, but also in other parameters (e.g. relief, soil stoniness, historical development, type of colonization, population densities).

The coverage of commons based on aerial images was determined for six basic categories, which were combined according to a predetermined procedure. In particular, the recognition of categories “abandoned” and “afforested” in some cases was unclear. Sometimes it was difficult to identify whether the current vegetation cover is the result of spontaneous overgrowth or tree plantings. The actual condition could not be verified in the field in all cases. However, a field survey of a selected subset of locations revealed only a few errors. The identification of the type of cover from the images from the 1950s and 1990s was carried out according to the same methodology as for the current images. Poor image quality often made it difficult to identify the cover accurately. However, an important criterion that often offset this inaccurate classification in historical pictures was the preservation of the boundaries of the commons. The boundaries of

former commons were, in contrast to the current situation, mostly well distinguishable in images from the 1950s and 1990s.

The significant increase in all combinations of cover between 1950s–1990s and 1990s–2019 is not to be overlooked and was partly expected. This phenomenon is based on the general increasing tendency of land cover to change due to socio-economic changes occurring at that time.

An important period for significant changes in the use of commons was the collectivization of agriculture, which started in 1948. After this process, melioration occurred mainly in the 1960s–1970s. Many locations, valuable from the conservation point of view, however, were drained and converted to agriculture lands even during the second half of the 1980s. As reported by Luka et al. (2017) large-scale drainage significantly changed the Czech landscape.

Its main purpose was to expand the agricultural area and increase food production. The tendency towards food self-sufficiency thus caused the amelioration of a significant part of the landscape, whose functions had so far been other than just production. The commons could be an example of a part of the landscape, which have lost their mosaic and overall biological and cultural value. Commons used to be rich wetland localities, but due to amelioration, they were drained, ploughed and converted into fields. The same trend, i.e. converting of meadows and commons to arable land in the second half of the 20th century, is also mentioned by Kaninska et al. (2014), who examined changes in the landscape in the Slovak foothills. After the 1950s there was almost a 20% increase in the number of abandoned commons. Many of them were abandoned already in 1920s or 1930s; however, successional changes were not too apparent during the first decades. Similarly, some commons recognized as abandoned during the socialist era (1950s–1990s) were in fact unmanaged already before or during WWII. They were not recognised as abandoned based on aerial photographs taken in the early 1950s because the successional changes were not recognizable at that time. Natural succession is usually slower and less apparent during the first years after the ending of management and overgrowth accelerates in the later stages of succession (e.g. Joyce 2014). In the area studied, this common trend was supported by social-economical changes during the communist era, when private and municipal ownership of the land completely disappeared and many chaotic measures escalating in succession could happen on abandoned commons (e.g. litter of old hay or manure, municipal waste landfills, irregular cutting of firewood etc.).

The high percentage of abandoned commons after 1990s can be explained by socio-economic changes during the last 30 years. Although these areas were returned to their original owners in the 1990s, not all subsequent owners had the tools, capacities, finances and

will to restore the long-term abandoned former commons. Many of them continued to be uncultivated and succession continued. The speed of succession and the current amount of woody vegetation on these former commons was influenced by a number of factors, including both abandonment, previous management, local ecological conditions (soil, humidity, nutrient availability, etc.), diaspora source, various disturbances (game activities, casual visitors), change in techniques, etc. (Benjamin et al. 2005).

Answering the question raised by Hardin's essay (1968), we can say that the history of commons is not a complete tragedy, notwithstanding that our inventory revealed that three quarters of former commons have disappeared for different reasons. There is still, however, a great opportunity to save the rest of them and benefit from these treasures in our landscape. The existence of these extensively managed areas in our humdrum landscape is very important. This type of ecosystem provides many services. In contrast to intensive grazing, which forms the main part of the homogenized landscape, commons contribute to the preservation of species biodiversity, provide natural refuges for specific species and, among other functions, significantly help retaining water in the landscape. They significantly contribute to mitigating climate change and support sustainable landscape management. The cultural and historical significance for the local people and aspects of human well-being are also worth highlighting.

There are mosaics of significantly valuable habitats, in which populations of rare and protected species occur on all of the currently preserved commons. Some of them are managed for their conservation value: there is a nature reserve and several localities with endangered species, the management of which is paid from natural conservation funds. Local farmers or members of hunting clubs occasionally manage several others. However, many of the currently preserved commons lack appropriate management. More detailed evaluations of their conservation value and appropriate management are very much needed. More preserved and eventually restored commons should be included in the Territorial system of landscape ecological stability (i.e. ecological network), because they can play an important role as biodiversity stepping stones and improve the structure of the landscape.

Acknowledgements

We thank the following institutions and persons: A. F. G. Dixon for revising the language and J. Koreš for his endless and contagious enthusiasm for commons. This study was funded by the Ministry of Education, Youth and Sports of CR within the CzeCOS program, grant number LM2018123.

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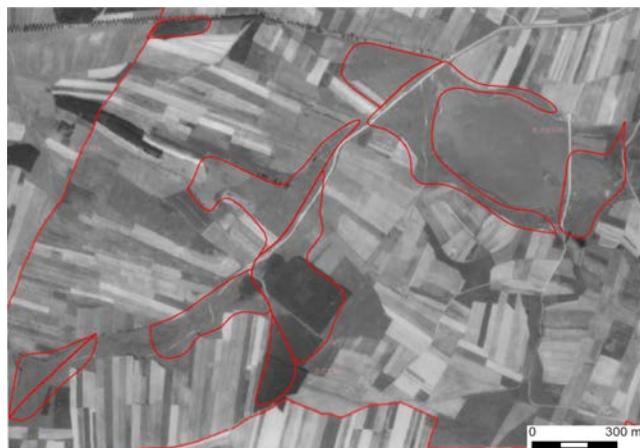
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Appendix 1

A series of pictures showing changes over time in Malý Bor cadastre – an illustrative segment of the area studied. Red line – borders of common pastures.



1850s



1950s



1990s



2019

FACTORS DETERMINING THE DISTRIBUTION OF ORCHIDS – A REVIEW WITH EXAMPLES FROM THE CZECH REPUBLIC

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ABSTRACT

The natural environment has been significantly altered by human activity over the past few decades. There is evidence we are now experiencing the sixth mass extinction, as many species of plants and animals are declining in abundance. We focused on the Orchidaceae because this plant family has experienced one of the biggest reductions in distribution. We investigated patterns in species richness and distribution of orchids, the rate and causes of their decrease and extinction, and factors influencing their occurrence in the Czech Republic and Greece. The key findings are: (i) Method of pollination and type of rooting system are associated with their distributions and they are different in the two countries. We assume that these differences might be due to the difference in the orography, distribution of suitable habitats and types of bedrock in these two countries. (ii) The greatest reduction in distribution was recorded for critically endangered taxa of orchids. The number of sites suitable for orchids in the Czech Republic declined by 8–92%. The most threatened orchid species are *Spiranthes spiralis*, *Anacamptis palustris*, *Epipogium aphyllum* and *Goodyera repens*. The distribution of orchids in the Czech Republic is mainly determined by the distribution of their habitats. (iii) The most important factor affecting the distribution of Czech orchids in South Bohemia is land cover. And the most important types of habitats (types in KVES) are oak and oak-hornbeam forests and agricultural meadows. Based on this information, it should be possible to improve the management that is crucial for maintaining orchid localities.

Keywords: decline; environmental factors; extinction; Maxent; orchids; pollination; root system

Introduction

Worldwide biodiversity is currently decreasing dramatically. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), working under the UN auspices, published an extensive report on plant and animal biodiversity in May 2019. According to this report, we are facing the sixth global extinction of species with species diversity decreasing worldwide at a fast pace, the rate of species extinction is now a hundred times greater than the average for the last ten million years and one-eighth of existing species are endangered (<https://ipbes.net/global-assessment>). Furthermore, the report of IPBES states that approximately three quarters of the terrestrial and two thirds of the marine environment have been significantly altered by human activity. One of the main reasons for this huge decrease in biodiversity in the world is loss of the natural habitats of plants and animals (<https://ipbes.net/global-assessment>).

Orchids are known all over the world because of their beautiful flowers in the wild, as well as in our gardens and homes, and have become very popular in the last few decades. There are many publications on the distribution of orchids worldwide, which indicate that both professionals and the lay public are interested in orchids (e.g. Millar 1978; Seidenfaden and Wood 1992; Bose et al. 1999; Dykyjová 2003; Vlčko et al. 2003; Jersáková and Kindlmann 2004; Průša 2005; Averyanov et al. 2015; Antonopoulos and Tsiftsis 2017; Grulich 2017; Tsiftsis and Antonopoulos 2017; Kühn et al. 2019; Knapp et al. 2020;

Wagensommer et al. 2020 and many others). Unfortunately, the family Orchidaceae is one of the most threatened plant families with a high risk of species extinction (Swarts and Dixon 2009). Orchids are disappearing worldwide, mostly due to habitat loss, but other factors like climate change are likely to increase in importance during the 21st century (Wotavová et al. 2004; Pfeifer et al. 2006). Because of the high risk of extinction, orchids are listed in CITES and protected by law in many countries.

Despite the high number of studies on orchids, we still lack critical information necessary for the conservation of Orchidaceae, especially for species that are known to be threatened or endangered. All aspects that will be mentioned below make orchids an excellent plant family for various studies on various aspects of biology.

Orchids and their Specialized Life Strategies

The orchid family is an important group with respect to conservation biology (Pillon and Chase 2006), because so many are threatened with extinction (Swarts and Dixon 2009). Many characteristics, such as great species richness, specific role in ecosystems, or threat of extinction, make it crucial to explore the distribution and conservation status of Orchidaceae (Zhang et al. 2015).

Orchids, with approximately 28 500 species (Gov-aerts 2020) are the most diverse and widespread family of flowering plants (Swarts and Dixon 2009) and are classified among the most threatened groups worldwide (Cribb et al. 2003; Kull and Hutchings 2006). They are an

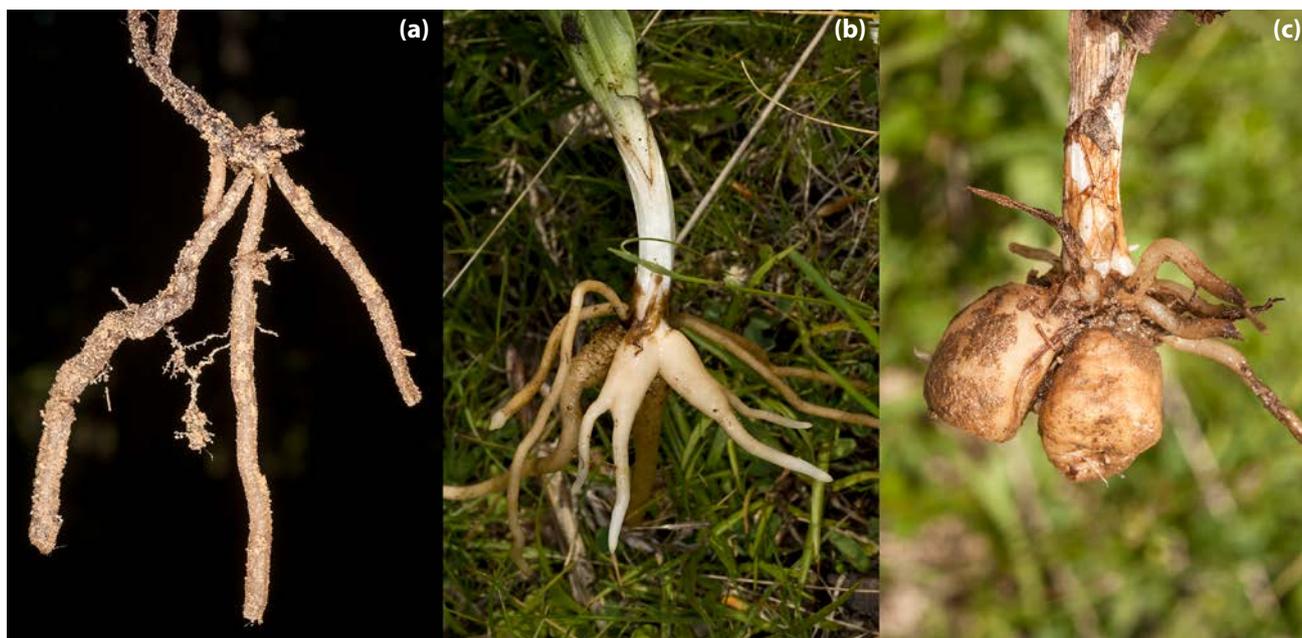


Fig. 1 Different types of orchid rooting systems: (a) rhizomatous, (b) intermediate and (c) tuberous.

ideal group for exploring determinants of species diversity because they are well recorded and studied in many countries in Europe (Kull et al. 2006).

Most species of orchids are threatened in the wild (Cribb et al. 2003) and are disappearing from their natural habitats worldwide (Cribb et al. 2003; Kull and Hutchings 2006; Knapp et al. 2020; Wagensommer et al. 2020). In Europe, all orchids are terrestrial and can be found in almost all habitats (Hågsater and Dumont 1996; Delforge 2006; Štípková et al. 2017). The most species-rich area in Europe is Southern Europe, especially the Mediterranean area (Del Prete and Mazzola 1995; Hågsater and Dumont 1996). Certain orchid genera (e.g. *Ophrys*, *Serapias*), for which the Mediterranean area is a centre of evolution, are remarkably species diverse (Del Prete and Mazzola 1995; Phitos et al. 1995; Pridgeon et al. 2001), whereas the greatest species diversity of species-rich genera are of more northern origin (e.g. *Epipactis*, *Dactylorhiza*) is recorded in central and northern Europe (Averyanov 1990). The availability of detailed records provides opportunities for comparative analyses of the declines in species over time.

Therefore, it is a pity that despite the high number of studies dealing with orchids, we still lack rigorous analyses of this data aimed at determining the relative importance of environmental factors and species traits associated with the decline in the numbers of sites suitable for orchids and particular species. However, such an analysis is crucial for their conservation in terms of an effective management of orchid sites (Kull and Hutchings 2006). Terrestrial orchids are probably one of the best examples of the decline in biodiversity in plants.

There is an important life history trait that plays a significant role in determining orchid presence/absence

and distribution in space: their **rooting system**, which is thought to represent particular strategies for underground storage of resources (Rasmussen 1995). In some species, the rooting system consists of a simple rhizome, whereas in others it is thicker and tuberous and serves as a storage organ. Among the European orchids, the genera *Epipactis*, *Cephalanthera* and *Cypripedium*, which are believed to be the most primitive, have short rhizomes. The most important evolutionary development in the growth forms of Orchidaceae was the production of efficient storage organs (tuberoids). In this evolutionary process, *Pseudorchis albida* is the most primitive tuberoid orchid, whereas the palmate tuberoids (*Dactylorhiza*, *Coeloglossum*, *Gymnadenia*) and those with fusiform tubers (e.g. *Platanthera*) evolved later (Dressler 1981; Averyanov 1990; Tatarenko 2007). Coarse division of the European orchids in terms of their rooting systems could be useful for testing hypotheses on their patterns of distribution, as this trait has evolved and differentiated in response to changing climatic conditions (Averyanov 1990).

Following the evolutionary trends in temperate orchids (Dressler 1981; Averyanov 1990; Tatarenko 2007), the species of orchids were classified here in three categories based on the above-mentioned morphology of their root system, which also indicates how primitive or highly evolved an orchid is. Based on this classification, the first species group consists of the rhizomatous orchids (*Cephalanthera*, *Corallorhiza*, *Cypripedium*, *Epipactis*, *Epipogium*, *Goodyera*, *Hammarbya*, *Limodorum*, *Liparis*, *Malaxis* and *Neottia*), the second, those with palmate or fusiform tubers, which is the intermediate stage (hereafter referred to as intermediate) in the evolution of temperate orchids in Eurasia, and includes species of the genera *Dactylorhiza*, *Gymnadenia*, *Platanthera* and *Pseudorchis*. The third

species group consists of those orchids with a spheroid or spindle-shaped tuberous root system (*Anacamptis*, *Herminium*, *Himantoglossum*, *Neotinea*, *Ophrys*, *Orchis*, *Spiranthes* and *Traunsteinera*). See Fig. 1 for illustrations of these categories.

Relationship between species richness of orchids with different rooting systems and various ecological factors and degree of specialization based on specific environmental conditions have not been previously studied in Europe. To fill this gap in our knowledge, we explored the associations of orchid species richness and the degree to which an orchid species is adapted to living in specific environmental conditions (in terms of species specialization index) with altitude in the Czech Republic (Štípková et al. 2021a) and with various ecological factors in Greece (Tsiftsis et al. 2019).

In addition to the differences in their rooting systems, orchids have very complicated **pollination strategies**. Survival of an orchid population or even a species may strongly depend on pollination and subsequent seed production (Jacquemyn et al. 2005a). As specialized pollination systems may be particularly vulnerable to anthropogenic modification of landscapes (Anderson et al. 2011; Pauw and Bond 2011; Phillips et al. 2015) and may strongly affect species survival.

Generally, orchids are characterized by a diversity and specificity of pollination mechanisms, which may involve the food-foraging, territorial defence, pseudoantagonism, rendezvous attraction, brood-site and shelter imitation, sexual response, or habitat-selection behaviour of their pollinators (Ackerman 1986; Tremblay 1992; Tremblay et al. 2005; Jersáková et al. 2006; Micheneau et al. 2009). Most plants pollinated by animals produce and offer rewards to attract pollinators to visit their flowers (nectariferous species; Simpson and Neff 1983). Nectar is the most common floral reward (Dressler 1981; Jersáková and Johnson 2006) and can influence several aspects of pollinator behaviour (Jersáková and Johnson 2006). However, some plants attract pollinators, but do not offer any reward (nectarless – often also called deceptive – species; Heinrich 1979; Bell 1986). The nectarless strategy has evolved in many plant families, but most nectarless species are orchids (Renner 2005; Jersáková et al. 2006). In general, plants of nectariferous species are visited more frequently than nectarless plants (Neiland and Wilcock 1998; Pellissier et al. 2010). Pollinators also visit more flowers per inflorescence of nectariferous than nectarless species (Jersáková and Johnson 2006; Hobbhahn et al. 2017). Nectariferous species are less pollinator-specific than nectarless species, among which the most pollinator-specific are sexually deceptive species (Cozzolino and Widmer 2005; Phillips et al. 2009). As many as 60–70% of orchids have a single species of pollinator (Tremblay et al. 2005). This level of specialization (Tremblay 1992; Phillips et al. 2009) makes orchids vulnerable to fluctuations in pollinator abundance. Nectariferous orchids are better competitors for pollinators

than nectarless orchids (Pellissier et al. 2010). All this has consequences for fruit production and the fitness of the plants. As a result, nectariferous species have a higher fruit set than nectarless ones (Neiland and Wilcock 1998; Tremblay et al. 2005; Phillips et al. 2009; Hobbhahn et al. 2017) in all geographical areas (Neiland and Wilcock 1998) due to pollination limitation (Neiland and Wilcock 1998; Tremblay et al. 2005). Based on the above, we propose that pollination strategy plays a role in orchid distribution (Štípková et al. 2020b).

All the above and a range of ecological conditions affect the altitudinal and spatial distribution of orchids. For example, on La Reunion Island, Jacquemyn et al. (2005b) report that animal-pollinated orchids are more abundant at lower altitudes, whereas at high altitudes orchids tended to be auto-pollinated and cleistogamous. In Switzerland, the relationship between altitude and frequency of orchids with different reward strategies indicates a significant decrease in the occurrence of nectarless species of orchids with increase in altitude (Pellissier et al. 2010).

In addition to the pollination strategy, pollinator abundance can also affect fruit set in orchids. Pollinator abundance is influenced by the climate (temperature, seasonality) in a given area, which in turn is strongly determined by altitude (Arroyo et al. 1982; Körner 2007). Although the testing of the associations of species richness and niche breadth with altitude are frequently referred to in the literature (e.g. Kluge and Kessler 2011; McCreadie et al. 2017; Herrera et al. 2018; Vargas et al. 2008 and so on), none of these studies distinguish between pollination strategies (nectariferous/nectarless).

Mycoheterotrophy allows orchids to adapt to a wide variety of habitats, even those with extreme conditions (e.g. sites with little soil or lack of light). In the upper mountain zone, although it rains equally all year round the upper soil horizons are rich in organic matter (mostly in forested habitats), orchids (mostly rhizomatous and to a lesser extent palmate or fusiform tuberoids) are adapted to the low light conditions, often involving obligate mycoheterotrophy (Jacquemyn et al. 2017). The tuberous orchids mostly occur in open, dry and hot environments around the Mediterranean and in nutrient poor and eroded soils (Averyanov 1990; Delforge 2006). Although in these areas, low availability of soil water and nutrients are causes of stress (contrary to light, which is the cause of stress in forested habitats), fungi provide orchids with the water and nutrients necessary for their survival and growth. Moreover, when conditions (e.g. climatic) are unsuitable, the underground organs of orchids can remain alive and dormant, exploiting fungi, for several years (Rasmussen 1995; Shefferson et al. 2018).

Orchids and their Conservation

One of the key goals of conservation is to determine what causes declines in biodiversity and suggest ways

of stopping or slowing it down (Gaston and Blackburn 2000). This is especially true in Europe, where the numbers of species, abundances and distributions of many species of plants and animals have dramatically declined during recent decades.

The need for effective conservation measures is urgently required for areas and countries that were affected by human activities in past decades, and thus have lost a part of their biodiversity or the distributions of certain species have been greatly reduced (Štípková and Kindlmann 2021; Štípková et al. 2021b). It is commonly accepted that urbanization, land use changes and intensification of agriculture have resulted in a dramatic loss and fragmentation of habitats (Stewart 1992; Fischer and Stöcklin 1997; Kull et al. 2002, 2016; Bilz et al. 2011; Tsiftsis et al. 2011). The current landscape in Europe is mainly a result of recent changes in farm management (Henle et al. 2008). This affected the composition of the flora and fauna in most areas and resulted in a decline in European biodiversity (Fahrig et al. 2011; Ferreira et al. 2013; Brunbjerg et al. 2017; Fardila et al. 2017; Poschlod and Braun-Reichert 2017; Hass et al. 2018; Kurze et al. 2018). As for most other taxonomic groups, the reasons for the decline in orchid biodiversity include habitat loss, eutrophication and fragmentation (Wotavová et al. 2004; Janečková et al. 2006; Kull and Hutchings 2006; Kull et al. 2016). Central European countries have been intensively affected by changes in land use or agricultural intensification. Among these countries, the Czech Republic was strongly affected by such changes during the last few decades (Štípková et al. 2021b). In the past, there were important changes in the use of land in the Czech Republic, which differed from those that occurred in western parts of Europe due to changes in the political regimes (Adams and Adams 1971; Wädekin 1982; Krčmářová and Jeleček 2017). Before 1948, fields and meadows were traditionally managed (Krčmářová and Jeleček 2017), which involved mowing and grazing, low intensity agriculture of small fields and low application of fertilizers (Adams and Adams 1971). After 1948, small fields were consolidated into huge fields (Skaloš et al. 2011) and subsidies for fertilizers were provided, which resulted in high levels of nutrient chemicals in the soil (Adams and Adams 1971). As a result, many orchids declined and can now only be found at a small number of sites (Štípková and Kindlmann 2021). After the change in regime in 1989, the subsidies for fertilizers ceased, which resulted for a while in a great decline in the use of fertilizers (Reif et al. 2008). The implications for the survival of sites suitable for orchids, however, were not dramatic (Štípková and Kindlmann 2021).

Knowledge of orchid ecology, including environmental gradients that influence the patterns in orchid abundance, distribution, richness and composition, is essential for planning and applying conservation strategies and actions (Tsiftsis et al. 2008; Swarts and Dixon 2009), and lack of such knowledge negatively affects our ability to

identify sites that are worth protecting. We also still lack the knowledge needed to develop management plans for orchids under current or future scenarios of habitat loss and climate change.

Among others, there are two crucially important values when orchid conservation and survival during climate change is considered: number of species per unit area and the degree to which an orchid species is specialized to specific environmental conditions. The former clearly determines the conservation value of an area, while the latter tells us how much a species may be endangered by changes in environmental conditions, e.g., climate change. Both values were used for assessing the factors that affect the distribution of Czech orchids (Tsiftsis et al. 2019; Štípková et al. 2020a; Štípková et al. 2021a).

Patterns in the Distribution of Orchids

Understanding the abundance and patterns in the distributions of species at large spatial scales is one of the key goals of biogeography and macroecology (Gaston and Blackburn 2000; Tsiftsis et al. 2019), but effective conservation requires knowledge of species at small spatial scales (Tsiftsis et al. 2008; Swarts and Dixon 2009).

Species richness decreases from the equator towards the poles (Crame 2001; Francis and Currie 2003) and this pattern is among the most consistent in biogeography (Hillebrand 2004). The dependence of species richness on altitude is usually hump-shaped (Vetaas and Grytnes 2002; Bhattarai and Vetaas 2003), or monotonically decreases with increasing altitude (Bachman et al. 2004; Jacquemyn et al. 2005b), but sometimes species richness increases with altitude or shows an inversely unimodal trend; more rarely there is no obvious trend (Grytnes 2003; Hrivnák et al. 2014). In temperate regions, plant species richness is lower in areas that are cold compared to those that are warm, while species niches and range sizes tend to be broader (Stevens 1989; Thompson 2005). However, in addition to environmental gradients, there are other important factors that influence these patterns and niche breadth, e.g. the life-history strategies of species (Kostikova et al. 2013). Global warming has a direct effect on species distributions, as over the last few years there has been an increase in the number of species of plant species occurring in high mountains in Europe (Steinbauer et al. 2018). Although distributions of some species now extend further north or to higher altitudes than previously, other species are becoming more restricted due to the desertification observed in the southern parts of Europe (Karamesouti et al. 2015).

Species distribution models (SDMs) are a useful tool, which over the last few decades were often used in many branches of biogeography, conservation biology and ecology (Elith and Leathwick 2009), especially in stud-

ies on threatened species (Guisan et al. 2013). These numerical tools combine species occurrence records with environmental data (Elith and Leathwick 2009). In combination with GIS techniques, these models are especially important and useful for predicting the occurrence of rare species (Guisan and Thuiller 2005). Although the results of species distribution models often suffer from high levels of uncertainty due to biases in species distribution data, errors in environmental variables used as predictors, spatial resolution and the modelling process (Elith and Graham 2009; Rocchini et al. 2011), SDMs are nevertheless widely used to predict species distributions (Tsiftsis et al. 2012).

The maximum entropy algorithm in the MaxEnt application (Elith et al. 2006; Phillips et al. 2006; Phillips and Dudík 2008; Elith et al. 2011) is often used for modelling species distributions based on presence-only species records (Elith et al. 2011). This approach is used by conservation practitioners for predicting the distribution of a species from a set of occurrence records and environmental variables (Elith et al. 2011; Fourcade et al. 2014). MaxEnt is one of the most robust methods in terms of successfully estimating the area of distribution from only a few records of occurrence (Hernández et al. 2006; Yi et al. 2016). Despite the long history of studies on orchids, very few of the previous papers on the distribution, phytogeography, or conservation strategies for orchids are based on using species distribution models (e.g., see Kolanowska 2013; Wan et al. 2014; Reina-Rodríguez et al. 2016; Vollering et al. 2016). Presence-only modelling methods require a set of known species occurrences together with predictor variables, such as, topographic, climatic, edaphic, biogeographic, and/or remotely sensed data (Phillips et al. 2006; Phillips and Dudík 2008; Štípková et al. 2020a).

Factors Affecting the Distribution of Orchids

Questions concerning species diversity have attracted ecologists for over a century. Recently, this issue became even more important, because the diversity of life on Earth is in rapid decline (Dirzo and Raven 2003). Therefore, one of the most pressing tasks facing the global conservation community is trying to understand the main factors determining the diversity of species (Possingham and Wilson 2005) and identifying important areas for conserving biodiversity (Tsiftsis et al. 2011). Orchids are also known to be affected by environmental changes (Dirzo and Raven 2003), as well as to their high risk of extinction, compared to other plant families, as a result of natural and/or anthropogenic causes (Hutchings 1989; Kull et al. 2006).

One of the most worrying issues is that we still do not know the optimal abiotic and biotic requirements for population persistence of many species of orchids (Swarts and Dixon 2017). There are only a few studies in

the Czech Republic dealing with the factors that determine orchid presence/absence and distribution in space, and most of them include only one or a few species and/or a limited part of the distribution of the species studied (e.g. Štípková et al. 2017, 2018).

On a regional scale, geological substrate and the distribution of suitable plant communities determine the distribution of species (Tsiftsis et al. 2008), whereas on broad geographical scales, plant species richness is largely determined by climatic conditions (Sanders et al. 2007; Acharya et al. 2011; Trigas et al. 2013), which are in turn mostly influenced by the altitude and latitude of the area studied.

A better understanding of how species richness, niche breadth and range size are associated with geographical and/or environmental gradients is of crucial importance for species conservation and may even help us predict the effects of global change, especially when considering the distribution of orchids (Swarts and Dixon 2009; Zhang et al. 2015). In spite of the many atlases of the distributions of orchids, there is only scattered information on the factors determining orchid distribution and species richness throughout the Czech Republic (Štípková et al. 2020a; Štípková et al. 2021a).

Conclusions

In this review, we present a new insight into facts that affect orchid life. Although the majority of the studies are for the Czech Republic, we believe that our results and suggestions are also applicable to other parts of Central Europe, as well as other temperate regions.

The distribution of orchid taxa with different rooting systems and pollination strategies in the Czech Republic strongly depends on the distribution of suitable habitats and types of bedrock, together with mycorrhizal fungi, at different altitudes in the country. The association of altitude with the richness of orchid flora in the Czech Republic is much stronger than that with biogeography. On the contrary, the patterns in the distribution of Greek orchid taxa with different rooting systems are associated with geology and the special topography (particularly in terms of altitude, latitude and climate) as well as with the biogeography of the area.

The distributions of many species have decreased markedly over time. We assume that these changes are directly associated with changes in agriculture practices in the Czech Republic and abandonment of traditional management. We suggest that authors should use the most precise spatial resolution available in order to avoid misinterpretation of their results. We found that the vast majority of orchids have disappeared from many of their historical localities and four orchids became extinct. The most threatened orchids in the Czech Republic are *Spiranthes spiralis*, *Anacamptis palustris*, *Epipogium aphyllum* and *Goodyera repens* (Štípková and Kindlmann



Fig. 2 Photographs of the most threatened species of orchids in the Czech Republic: (a) *Spiranthes spiralis*, (b) *Anacamptis palustris*, (c) *Epipogium aphyllum* and (d) *Goodyera repens* (a, c, d © Z. Štípková; b © J. Štěpán).

2021, Fig. 2). All these changes seem to be closely associated with changes in agricultural practices and in the use or alteration of orchid natural habitats. We believe that these results can be used to set up specific conservation measures that are needed either to prevent further decline in orchids or the recovery of specific orchid populations.

The most important factor that affects the distribution of many orchids in the South Bohemian region of the Czech Republic is land cover. Thanks to potential distribution maps, we found other places with suitable environmental conditions for orchids. These findings may help the conservation of orchids by protecting those habitats with suitable environmental conditions.

Acknowledgements

We thank the Nature Conservation Agency of the Czech Republic for giving us permission to use their dataset. We are greatly indebted to Tony Dixon for helpful hints on how to improve the style of English in this paper.

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PHYTOREMEDIATION OF SOILS POLLUTED WITH HEAVY METALS IN THE VICINITY OF THE ZENICA STEEL MILL IN BOSNIA AND HERZEGOVINA: POTENTIAL FOR USING NATIVE FLORA

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ABSTRACT

This study investigates levels of soil pollution and estimates the phytoremediation potential of 7 native plants growing close to the Zenica steel mill. Plant leaves or roots and associated soil samples were collected from this site and characterized in terms of the concentrations of the heavy metals Cr, Cd, Pb, Zn, Cu and Ni. Heavy metal concentrations in soil and plant samples were determined using atomic absorption spectrophotometry. Bioaccumulation factors for heavy metals were also calculated. All plants studied had a low ability to remove or stabilize heavy metals in soil. This was probably associated with the poor mobility and thus poor availability of heavy metals to the plants growing in the vicinity of this steel mill.

Keywords: bioaccumulation factor; leaves; roots; soil properties

Introduction

High concentrations of heavy metals in soils can damage ecosystems and consequently human health (Fazekašová and Fazekaš 2020). Therefore, methods of preventing and for restoring soils polluted with heavy metals are needed. There are some conventional ways of restoring contaminated soils, which are based on physical and/or chemical methods, including soil washing, membrane filtration, chemical precipitation, etc. Despite being widely used, these methods have two main disadvantages: the chemicals and/or physical treatments used cause significant changes in the physicochemical and biological characteristics of soils and are very expensive (Bradl and Xenidis 2005; DalCorso et al. 2019).

Phytoremediation is a recently developed technology that reduces or stabilizes heavy metals in soils using plants (Yan et al. 2020). It is defined as plants' or root-microorganisms' ability to remove, stabilize, degrade or isolate toxic substances from the environment. Among the different phytoremediation techniques, phytoextraction and phyto stabilization are the most widely used for the rehabilitation of heavy metal polluted soils because these techniques are low cost, environmentally friendly and thus more acceptable to the public (Rai 2008; Zgorelec et al. 2020). Phytoextraction is the absorption of heavy metals by roots followed by their translocation and accumulation in the aboveground parts of plants. Plants that produce high levels of biomass grow rapidly, are easily

cultivated and harvested, and most importantly, tolerate and accumulate high concentrations of heavy metals in the aboveground parts are considered to be appropriate for phytoextraction (Zhou et al. 2018). On the other hand, phytostabilization is the ability of plants to reduce the mobility of heavy metals via absorption and accumulation by roots, adsorption onto roots or by changing the solubility of heavy metals by means of root exudates. Plants native to the polluted soil that can retain large quantities of metal ions in their roots or through root exudates lower their solubility are considered to be appropriate for phytostabilization (Monaci et al. 2020). Although phytoremediation is not consistently effective, it is undoubtedly causes less damage to the environment than the use of chemicals and/or physical methods (Azubuike et al. 2016).

Unfortunately, most soils in the central and north-eastern parts of Bosnia and Herzegovina are polluted with heavy metals due to industrial, mining or agriculture. This problem is particularly evident in the Zenica region where industrial activity is constantly increasing. Although the soils in this area are to a greater or lesser extent polluted with heavy metals many plants successfully grow in these soils and, therefore, these plants could be very interesting candidates as potential phytoaccumulators or phytostabilizers.

The objectives of this study were to identify the native flora growing in heavy metal polluted soils near the Zenica steel mill and evaluate the ability of some of them to remove or stabilize heavy metals in polluted soils.

Materials and Methods

Study area

Zenica steel mill is located in the city of Zenica (44°13'8" N, 17°53'16" E), in the Bosnia river valley, about 70 km northwest of Sarajevo. The climate in this area is warm and temperate. According to Köppen and Geiger, this climate is classified as Cfb. The average annual temperature in Zenica is 11.3 °C and rainfall is 952 mm. The rainfall in Zenica is significant, with precipitation even during the driest month.

Three soil plots were studied in the immediate vicinity of the Zenica steel mill. All of the plots were located at a distance of about 300 m from the steel mill in a south-easterly direction. The plots were approximately 500 m² in area and located within 200 m of each other. According to Word Reference Base for Soil Resources, the soils are classified as Leptosols (IUSS 2015). Leptosols develop on limestone, marble, dolomite and other carbonate-rich parent rocks, and as a result are neutral or slightly alkaline. Furthermore, Leptosols are well-drained soils with good aeration, but poor moisture retention; however, the capacity of these soils to promote plant growth and development is very diverse and depends primarily on the depth of the soil profile and the type of carbonate bedrock.

Flora in the area studied

Vegetation surveys were conducted in three plots measuring 5 × 25 m using a quadrat sampling method to determine the composition of the flora. The high density and cover of native species of plants were the criteria for plot and quadrat selection. D.A.F.O.R scale (Dominant, Abundant, Frequent, Occasional and Rare) was used for measuring species' abundance, while plant density was determined by counting the number of individual plants of a species and dividing it by the quadrat's area. A total of 71 species of plants was identified. Also, the vegetation surveys revealed the presence of a large number of native plants belonging to the families *Poaceae* (12), *Fabaceae* (9), *Asteraceae* (7), *Chenopodiaceae* (5), *Brassicaceae* (4), *Caryophyllaceae* (4) and *Rosaceae* (4). In terms of density sweet clover (*Melilotus officinalis* (L.) Lam.) was present at the greatest density (sum of mature plant and seedling densities) (41.4 plants/m²) followed by mug wort (*Artemisia vulgaris* L.) (24.2 plants/m²), chickweed (*Stellaria media* L.) (21.3 plants/m²) and blueweed (*Echium vulgare* L.) (18.8 plants/m²). Accordingly, these plants were selected for evaluating their ability to remove heavy metals from polluted soils. Ground cover value was highest for hoary mullein (*Verbascum pulverulentum* Vill.) and therefore this plant was also evaluated for phytoremediation purposes.

Soil sampling and analysis

Soil samples were collected in February 2020 from the three plots. The samples were taken from a depth of

0–30 cm using a soil sampler probe. For each plot, the samples collected from five spots (north, south, east, west and center of plot) were thoroughly mixed to obtain a composite soil sample. The collected soil samples, each weighing ca. 500 g, were air dried, crushed and sieved through a 2 mm mesh. Soil pH was determined in H₂O and 1 M KCl solution (ratio soil/solution 1:2.5) using a pH meter, organic matter using the potassium dichromate redox method (ISO 1998) and available forms of phosphorus and potassium using the ammonium lactate (AL) method (Egnér et al. 1960).

Heavy metals in soil samples were extracted using aqua regia with a volume ratio of 1:3 HNO₃/HCl as follows: 1 g of air-dried, ground and sieved soil was accurately weighed in a 250 ml round bottom flask and digested with 21 ml of aqua regia under reflux on a hot-plate for 2 hours. Resultant solutions were cooled to room temperature, filtered through Whatman No. 42 filter paper and then diluted to 100 ml with deionized water (ISO 1995). Heavy metal concentrations in digested soil samples were determined using Atomic Absorption Spectrophotometry and a Shimadzu Atomic Absorption Spectrophotometer (AAS) model AA 7000. Calibration for each element was done using a series of standard solutions (Merck, Germany) and calibration graph with a correlation coefficient (r^2) > 0.999.

Plant sampling and analysis

Native herbaceous plants were collected during summer in 2020. For each species of plant, three samples were collected at random in the area where the vegetation was surveyed (quadrat). Each sample consisted of five plants that were carefully collected including as much of their roots as possible, then placed in paper bags and transported to the laboratory. Afterwards, these plants were carefully washed with distilled water, separately dried and ground, and then stored in paper bags.

Only samples of the roots of the native species of woody plants were collected during summer in 2020. Root fragments (<5 mm in diameter) from 5–15 cm deep were collected from five plants of each the selected species growing in the area studied. The root fragments were gently cleaned of soil particles, separately dried and ground, and then stored in bags prior to extraction.

Extraction of heavy metals from samples of plants was done as follows: 1 g of dried and ground plant material was placed in a 100 ml round bottom flask and then 10 ml HNO₃ and 4 ml of H₂SO₄ were added. The flasks were left for few hours at room temperature and heated gently on a hot-plate until light fumes were emitted. Then, the digest was cooled down to room temperature, filtered through a Whatman No.42 filter paper into 50 ml flask and diluted to the mark with deionized water. Heavy metal concentrations in these solutions were also determined using Atomic Absorption Spectrophotometry.

Bioaccumulation factor

The bioaccumulation factor (BAF) is defined as the ratio of heavy metal in the harvestable part of plants (aboveground biomass) to that in the soil in which they were growing. Accordingly, BAF was calculated using the following formula used by Kachenko and Singh (2006):

$$BAF = \frac{C_{shoots}}{C_{soil}} \quad (1)$$

where C shoots and C Soil are the heavy metal concentrations in the harvested aboveground plant material and soils, respectively. BAF values more than 1 indicate that the plant can tolerate or accumulate heavy metals (Petelka et al. 2019).

Statistical analysis

All measurements of plant samples were performed in triplicate and the results were presented as mean \pm standard deviation. The data were analyzed using One-Way Analysis of Variance (ANOVA) and the means compared using least significance difference (LSD) test at 5% significance level ($P < 0.05$).

Results

Heavy metal concentrations and basic chemical properties of the soils studied

Concentrations of heavy metals (Cr, Cd, Pb, Zn, Cu and Ni) in the soil in the plots are shown in Table 1.

The results indicate that concentrations of the toxic heavy metals Cr, Cd and Pb in the soil in the plots located near the steel mill greatly exceed the threshold established by Bosnia and Herzegovina legislation (OG FBiH 2009). Concentrations of the potentially toxic heavy metals Zn,

Cu and Ni also exceeded the thresholds prescribed by the same legislation, indicating that the soils close to the Zenica steel mill are polluted with heavy metals. In addition to heavy metal concentrations, the soil's basic chemical parameters were recorded (Table 2).

As shown in Table 2, all the soils studied were alkaline, with high CaCO_3 concentrations, moderate level of organic matter and relatively low content of available forms of phosphorus and potassium.

Concentrations of heavy metals in aboveground parts of plants

Concentrations of heavy metals (Cr, Cd, Pb, Zn, Cu and Ni) in the aboveground parts of native plants that grow abundantly in the soils studied are shown in Table 3.

For the 5 species of plants included in this study, the highest concentrations of heavy metals were recorded in the aboveground parts of *Verbascum pulverulentum* Vill. Concentrations of Cr, Cd, Pb, Zn and Ni in the aboveground parts of the other native plants studied, i.e. *Melilotus officinalis* (L.) Lam., *Echium vulgare* L., *Stellaria media* L. and *Artemisia vulgaris* L. were significantly lower. Moreover, concentrations of Cd, Pb, Zn and Ni in the above plants were lower or within the normal range of values for these elements in plants, indicating that these plants are unsuitable for phytoremediation purposes. The normal ranges of Cd, Pb, Zn and Ni in the leaves of plants are 0.01–2.4 mg kg^{-1} , 0.5–30 mg kg^{-1} , 20–100 mg kg^{-1} and 0.02–50 mg kg^{-1} , respectively (Chaney 1989). Interestingly, the concentration of Cu recorded in the aboveground parts of *Artemisia vulgaris* L. was the highest of the plants studied. This result was unexpected since the efficiency of this plant in removing other heavy metals from the soils was very low.

Table 1 Heavy metal concentrations in the soils.

Plot	Heavy metals (mg kg^{-1})					
	Cr	Cd	Pb	Zn	Cu	Ni
1	117.03	6.39	835.04	156.82	92.38	123.11
2	103.19	3.41	698.11	140.11	84.76	84.16
3	122.44	5.17	705.43	153.05	94.23	120.53
Limit value*	100.00	1.50	100.00	200.00	80.00	50.00

* Limit value prescribed by legislation in Bosnia and Herzegovina.

Table 2 Basic chemical properties of the soils studied.

Plot	Chemical properties					
	pH (H_2O)	pH (KCl)	humus (%)	Available P ($\text{mg } 100 \text{ g}^{-1}$)	Available K ($\text{mg } 100 \text{ g}^{-1}$)	CaCO_3 (%)
1	7.80	7.16	3.10	3.21	10.30	8.70
2	7.90	7.26	2.86	2.26	7.10	10.06
3	7.77	7.22	2.90	2.03	9.10	9.61

Table 3 Concentrations of heavy metals in aboveground parts of plants.

Plant	Heavy metals (mg kg ⁻¹)					
	Cr	Cd	Pb	Zn	Cu	Ni
<i>Verbascum pulverulentum</i> Vill.	7.45 ± 2.08 ^{a*}	0.28 ± 0.12 ^a	28.18 ± 8.20 ^a	71.98 ± 15.06 ^a	16.35 ± 5.05 ^c	4.03 ± 2.09 ^a
<i>Melilotus officinalis</i> (L.) Lam.	0.85 ± 0.20 ^d	0.07 ± 0.07 ^c	1.12 ± 0.84 ^c	26.61 ± 3.83 ^d	13.76 ± 6.56 ^c	0.38 ± 0.30 ^d
<i>Echium vulgare</i> L.	2.62 ± 0.89 ^b	0.03 ± 0.05 ^c	4.48 ± 1.56 ^b	49.34 ± 4.02 ^c	21.01 ± 4.03 ^b	3.11 ± 1.67 ^{ab}
<i>Stellaria media</i> L.	1.52 ± 0.88 ^{cd}	0.06 ± 0.04 ^c	1.86 ± 0.82 ^{bc}	26.47 ± 7.41 ^d	12.89 ± 3.33 ^c	0.68 ± 0.38 ^{cd}
<i>Artemisia vulgaris</i> L.	2.03 ± 0.37 ^{bc}	0.14 ± 0.12 ^b	3.73 ± 1.19 ^{bc}	63.32 ± 8.00 ^b	36.04 ± 5.12 ^a	1.59 ± 1.17 ^c
LSD _{0.05}	0.882	0.069	3.072	6.663	3.814	0.988

* Averages with the same letter are not significantly different ($P < 0.05$).

Table 4 Concentrations of heavy metals in the roots of plants.

Plant	Heavy metals (mg kg ⁻¹)					
	Cr	Cd	Pb	Zn	Cu	Ni
<i>Rhus typhina</i> L.	5.67 ± 3.99 ^{b*}	0.29 ± 0.17	25.01 ± 5.03	62.88 ± 5.34	16.06 ± 4.11	4.25 ± 3.11
<i>Populus nigra</i> L.	23.54 ± 4.06 ^a	0.46 ± 0.28	26.13 ± 3.39	63.79 ± 6.11	14.33 ± 5.02	3.19 ± 2.09
LSD _{0.05}	3.13	–	–	–	–	–

* Averages with the same letter are not significantly different ($P < 0.05$).

Table 5 Bioaccumulation factor values for heavy metal transfer from soils to plants.

Plants	Bioaccumulation factor					
	Cr	Cd	Pb	Zn	Cu	Ni
<i>Verbascum pulverulentum</i> Vill.	0.064	0.044	0.034	0.459	0.177	0.033
<i>Melilotus officinalis</i> (L.) Lam.	0.008	0.021	0.002	0.190	0.162	0.005
<i>Echium vulgare</i> L.	0.021	0.006	0.006	0.322	0.223	0.026
<i>Stellaria media</i> L.	0.015	0.018	0.003	0.189	0.152	0.008
<i>Artemisia vulgaris</i> L.	0.017	0.027	0.005	0.414	0.382	0.013
<i>Rhus typhina</i> L.	0.048	0.045	0.030	0.401	0.174	0.035
<i>Populus nigra</i> L.	0.201	0.072	0.031	0.407	0.155	0.026

Concentrations of heavy metals in the roots of plants

Concentrations of heavy metals (Cr, Cd, Pb, Zn, Cu and Ni) in the roots of the selected native woody plants growing in the soils studied soils shown in Table 4.

There are no significant differences in the concentrations of heavy metals in the roots of the woody plants studied except for Cr, which was significantly higher in roots of black poplar (*Populus nigra* L.) than staghorn sumac (*Rhus typhina* L.).

Bioaccumulation factor values for heavy metal transfer from soils to plants

Bioaccumulation factor (BAF) values for Cr, Cd, Pb, Zn, Cu and Ni for the plants studied area are shown in Table 5.

A total of seven species of plants consisting of five herbaceous plants and two trees had BAF values below 1 for each of the heavy metals, indicating that none of the plants is suitable for phytoremediation purposes.

Discussion

The present study found that the levels of Pb, Cr, Cd, Cu and Ni in the soils studied greatly exceeded the maximum allowable concentrations for agricultural soils. This is not surprising as all the soil studied came from plots situated close to the steel mill. Previous studies carried out in this area also indicate the soils there are polluted with toxic heavy metals (Bikić and Omerović 2012; Prčanović et al. 2012). Therefore, in order to reduce soil pollution caused by heavy metals in the vicinity of steel mills, more attention should be paid to the remediation of contaminated soils.

The present study focuses on the remediation of soils polluted with heavy metals using phytoaccumulation and Phyto stabilization. In order to achieve this goal, the present study evaluated the phytoaccumulation potential of five native herbaceous plants: *Verbascum pulverulentum* Vill., *Melilotus officinalis* (L.) Lam., *Echium vulgare* L.,

Stellaria media L., *Artemisia vulgaris* L. and the phyto-stabilization potential of two native woody plants: *Rhus typhina* L. and *Populus nigra* L. These plants grow in soils polluted with heavy metals without suffering any toxic effects, which is why they were included in this study. Several studies report that some of the above-mentioned plants can accumulate large amounts of heavy metals (Chandra et al. 2016; Gajić et al. 2018; Jakovljević et al. 2019), however, little is known about their ability to accumulate heavy metals when growing around the Zenica steel mill.

The BAF analyses revealed that these plants are efficient in accumulating Zn and Cu. BAF values for Zn, Cu, Cr, Cd, Ni and Pb, however, ranged from 0.189 to 0.459, 0.152 to 0.382, 0.015 to 0.201, 0.006 to 0.044, 0.005 to 0.035 and 0.002 to 0.034, respectively. This pattern of heavy metal accumulation in plants characterized by high bioaccumulation rates for Cu and Zn and low values for Cr, Cd, Ni and Pb is not surprising considering the importance of these elements for plant growth and development, as Cu and Zn are essential elements for plants and their presence in acceptable concentrations is crucial for maintaining their metabolic processes. Thus, plants have evolved mechanisms for the uptake and transport of both Zn and Cu (Bahamonde et al. 2019). On the other hand, Cr, Cd, Ni and Pb are highly toxic to plants even at very low concentrations, and therefore, many plants have developed different mechanisms for reducing their uptake and/or translocation. These mechanisms are mainly based on heavy metal compartmentalization in root cells, or biosynthesis of phytochelatin that bind with heavy metals and prevent their transport from roots to aboveground parts of plants (Anjum et al. 2015).

Interestingly, the lowest bioaccumulation rates recorded in this study were for Pb, suggesting that plants under Pb stress have mechanisms for reducing Pb uptake and translocation. Huang et al. (2017) report that cell walls of root cells restrict Pb uptake and thus act as an important protective barrier. That is, the Pb fixation by pectates in the walls of root cells prevents the entry of Pb into the internal tissues and its translocation within the plant.

The majority of studies report that ion homeostasis in plasmatic compartments is of great importance for the acclimation of plants to heavy metals stress conditions (Wiszniewska et al. 2019; Anwar and Kim 2020). Low toxic heavy metal ion activities in cytosol, nucleoplasm, mitochondria and other plasmatic compartments ensure regular functioning of metabolic processes and may be achieved mainly by vacuolar compartmentalization. Thus, in order to avoid heavy metal toxicity, plants accumulate toxic heavy metals in vacuoles, which indicate that vacuoles have a detoxification function. That is, non-hyperaccumulator plants largely accumulate toxic heavy metals in root vacuoles, while hyperaccumulator plants usually accumulate them in vacuoles in leaf cells following efficient long-distance transport. The different

strategies of non-hyperaccumulator and accumulator plants are associated with organ-specific differences in cell compartments, particularly in transition metal transporters (Sharma et al. 2016).

In view of the fact that all the herbaceous plants growing in the polluted soils in the vicinity of Zenica steel mill studied had relatively low heavy metal concentrations in leaves and low BAF values for toxic heavy metals, it is concluded that the above-mentioned plants cannot be regarded as hyperaccumulator plants. However, regardless of these findings, interestingly the concentrations of the hazardous heavy metals Cr, Cd and Pb were 5 to 15-fold higher in the leaves of *Verbascum pulverulentum* Vill., than in leaves of the other plants studied. These findings undoubtedly indicate that the ability of these plants to accumulate heavy metals depends, among other factors, on the plants' genetic background (Kozmińska et al. 2018).

Regarding BAF values, many scientists argue that a BAF > 1 does not necessarily indicate a plant is a hyperaccumulator (e.g., Robinson et al. 1998; Zhao et al. 2003). A value > 1 is unlikely to be recorded for plants growing on soils contaminated with heavy metals if the soil chemical and physical properties negatively affect the mobility of heavy metals and thus their availability to plants. That is, BAF is an effective way of assessing the mobility/availability of heavy metals in soils (Bempah and Ewusi 2016).

The results of this study also indicate that specific chemical properties of soils i.e. a high pH value and CaCO₃ content probably result in a significant reduction in the bioavailability of heavy metals, resulting in low BAF values and low accumulation of heavy metals in leaves. Many studies report that with increasing pH, organic matter and CaCO₃ content the mobility of most heavy metals in soils decreases due to their increased adsorption (Hamid et al. 2018; Palansooriya et al. 2020). In other words, in alkaline soils with high organic matter and CaCO₃ content, heavy metal ions tend to form insoluble hydroxides, carbonates and organic complexes and are unavailable to plants (Rieuwerts et al. 1998).

In this study, the Phyto stabilization potential of two native woody plants: *Rhus typhina* L. and *Populus nigra* L. was also evaluated. Unfortunately, the roots of both these plants have low ability to take up and accumulate heavy metals, which indicates that their potential for Phyto stabilization is low. This is probably due to the chemical properties of the soils, which is in accordance with the correlation between soil properties and the availability of heavy metals as described above.

Overall, the results of this study confirmed that the soils in the vicinity of the Zenica steel mill are contaminated with the hazardous heavy metals Cr, Cd, Ni and Pb. Furthermore, all the plants studied had a low ability to remove or stabilize heavy metals in the soil. This was probably due to the chemical properties of soils restricting the mobility of the heavy metals and thus their availability to plants. Thus, it is concluded that heavy metal uptake by plants is a complex soil-plant process, influ-

enced primarily by plant genetic background and soil physicochemical properties.

Acknowledgements

We are grateful for the financial support for this research from the Ministry of Education, Science and Youth of Sarajevo Canton.

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COMPARISON OF TEMPERATURE AND OXYGEN CONCENTRATION DRIVEN AERATION METHODS FOR BIODRYING OF MUNICIPAL SOLID WASTE

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ABSTRACT

In this study, we used a model bio drying reactor to optimize the drying of municipal solid waste (MSW). Two methods of aeration were used: one involved controlling the concentration of oxygen (16–20%) and the other the temperature in the upper part of the reactor (temperature 42–45 °C, oxygen concentration 16–20%). In terms of moisture content, the highest increase recorded was 0.94% and the highest decrease was 27.54%. The highest decrease in lower calorific value recorded was 9.23% and the highest increase was 41.12%. The energy balance in all the trial runs was positive. We noted that aeration using different concentrations of oxygen is strongly influenced by organic content. Thus, it is suitable only for drying wastes with known and stable compositions. The outcomes of the different methods used were influenced by ambient relative air humidity. Moisture gradients, which are often described in the literature, were not consistently reproduced in this study.

Keywords: aeration method; ambient moisture; bio drying; mechanical-biological treatment; municipal solid waste; organic content

Introduction

Bio drying is a process that utilises heat generated by aerobic decomposition of organic matter for drying substrates. The principal of the entire process lies in the air that is blown into the substrate, which is a source of oxygen for aerobic decomposition processes and a means of venting evaporated moisture. However, it also simultaneously cools the medium and negatively affects the energetic balance of the process. The main purpose of bio drying is to obtain a fuel with a high calorific value (Velis et al. 2009). Thus, it is necessary to determine optimal conditions for achieving a suitably dry substrate with the lowest energy consumption and lowest decomposition of organic matter. One of the main factors affecting this is the rate of aeration.

Rate of aeration was studied by Adani et al. (2002), who report that optimal drying efficiency can be achieved by using higher rates of aeration, which result in lower temperatures in the reactor chamber. Similar results are reported by Zhao et al. (2010), Cai et al. (2013), Yuan et al. (2018) and Zhang et al. (2020). The highest rate of aeration resulted in the highest evaporation, lowest decomposition of organic matter and highest calorific value of residue. The higher the rate of aeration, the greater the loss of moisture. However, when the substrate dries out, the microorganisms that decompose the organic matter suffer from a lack of water and become inactive (Walker et al. 1999; Adani et al. 2002; Avalos Ramirez et al. 2012). Huiliñir and Villegas (2014) report, that very high rates of aeration dries out and cools down the substrate and decreases the rate of decomposition of organic matter. Then the moisture content can be further decreased, but the convection drying effect is stronger than the bio drying effect. On the other hand, Vandergheynst et al. (1997)

report that when the rate of aeration is too low there is a deficit of oxygen in the upper parts of the substrate, which results in a decrease in the activity of microorganisms.

In addition, the studies using high rates of aeration also indicate there are gradients in temperature, moisture and calorific value in the reactor chamber (Adani et al. 2002; Sugni et al. 2005; Zhao et al. 2010), which results in the final substrate/fuel not being homogeneous in terms of quality.

The aim of this study was to compare two methods of aeration in order to determine the optimal rate. The two methods of aeration tested was one in which the oxygen concentration was controlled (as in industrial applications, e.g. Comptech biodrying system) and another in which temperature is controlled in the upper part of the reactor. Drying efficiency (decrease in moisture content, increase in calorific value), energy balance and occurrence of gradients recorded for these two methods were compared.

Material and Methods

Mixed solid waste (MSW) sample

A sample consisting of 1 m³ was collected from the regular MSW of the town Mníšek pod Brdy in the Czech Republic, which is the site of a proposed mechanical-biological waste treatment facility. Metal, glass and other rigid materials that could damage the equipment used in the subsequent grinding of the waste were removed. The sample was shredded using a grinder until the grain size was approximately 30 mm. The quartering method was used to obtain a 30 kg subsample for this study. This subsample was transported in a plastic barrel directly to the laboratory. A second 1 kg subsample was used for de-

Table 1 Composition of the substrate in the different treatments.

Treatment	Plastics (%)	Biodegradable (%)	Textile (%)
Temperature 1	26.4	59.7	13.9
Temperature 2	24.2	48.7	12.1
Oxygen 1	21.4	65.2	13.4
Oxygen 2	24.6	70.9	4.5

termining moisture content, calorific value and volatile matter content. Composition of the MSW sample is described in Table 1.

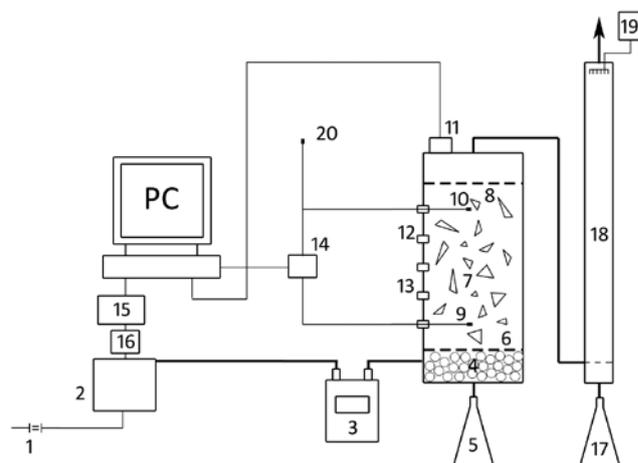
Model reactor

The reactor designed specifically for this study, consisted of a reactor chamber, a Secoh SLL 50 blower and a biofilter. The chamber of the reactor consisted of a 120 cm PVC cylinder with an internal diameter of 40 cm. A 3 mm mesh sieve was placed above the bottom of the reactor chamber to stabilise the waste and disperse the air supply. Below the sieve, there was a 10 cm layer of 5 mm glass beads to facilitate the removal of any leachate, which then drains out of the bottom of the reactor into an air-tight Erlenmeyer flask. The air inlet was above the bottom in the drainage layer and the exhaust vent in the lid of the chamber. The entire reactor was enclosed in a 10 cm thick layer of insulating material. Probes for sampling and measuring experimental conditions during the process were installed within the reactor walls. Papouch THT2 temperature and oxygen probes were placed in the upper and lower air passages. An ASEKO GTE oxygen sensor was placed under the reactor lid. In order to adjust the flow rate, we used an electronically-controlled blower. The reactor was connected to other parts of the system by plastic tubing. To determine the ambient temperature and humidity in the system, an external thermometer and hygrometer Papouch THT2 were attached to the reactor assembly about 0.5 m from the air inlet. These sensors were connected to a computer via signal converters. The blower was operated by a USB I/O module Papouch Quido 2/2. A multi-range BK G4 BO diaphragm gas meter was connected to the blower and the reactor chamber in order to measure the total volume of air that passed through the chamber (Fig. 1).

Aeration

The waste was dried using two methods of aeration: one by controlling the concentration of oxygen and the other by controlling the temperature in the upper part of the reactor.

During the run when the concentration of oxygen was controlled, the concentration in the upper part of the reactor was between 16% and 20%. When oxygen concentration fell below 16% or rose above 20%, the blower was turned on or off, respectively. According to Avalos Ramirez et al. (2012) an optimum oxygen concentration

**Fig. 1** Diagram of the reactor.

1 – Voltage regulator, 2 – Air source, 3 – Gas meter, 4 – Drainage layer, 5 – Erlenmeyer flask to drain leachate, 6 – Perforated sheet, 7 – Dried waste, 8 – Perforated sheet, 9 – Temperature and moisture meters, 10 – Temperature and moisture meters, 11 – Oxygen sensor, 12 – Port for sampling, 13 – Port for sampling, 14 – PC converter, 15 – USB I/O module for switching the air source, 16 – Relay for switching the air source, 17 – Erlenmeyer flask for collecting condensate, 18 – Biofilter (compost, peat, bark, expanded clay), 19 – The nutrient solution needed for the proper functioning of the biofilter (odour control), 20 – Ambient air temperature and moisture meters.

for aerobic degradation is between 15% and 20%. During drying, temperature and relative humidity were recorded in the upper and lower layer. The oxygen concentration in the upper part of the reactor was recorded, as were the ambient temperature and moisture outside the reactor. The oxygen concentration in the upper layer, the moisture and temperature were recorded at five second intervals and the activity of the blower was recorded at one second intervals. The measurements were taken with a Papouch THT2 sensor and ASEKO GTE oxygen sensor using Wix software. The data processing was done using Microsoft Excel and R.

During the run in which the temperature in the upper part of the reactor was controlled, the blower was turned on when the temperature in the upper layer reached 45 °C and switched off when it decreased to 42 °C. The upper temperature limit is based on the study of Adani et al. (2002). In addition, the oxygen concentration in the upper layer was maintained, as mentioned above, in the range 16% to 20% (Table 2).

Product collection and processing

There was four drying runs: two in which temperature was controlled and two in which the concentration of oxygen was controlled. The runs took from 253 to 259 hours. After every run, using the openings in the reactor, samples were taken from the upper and lower layer to analyse the drying efficiency. The weight of each sample was approximately 1 kg. Each sample was tested for moisture content, volatile matter content and calorific value.

Table 2 Conditions in reactor.

Run	Waste mass (kg)	Time (h)	Air consumption (m ³)	Air flow before (l/s)	Air flow after (l/s)
Temperature 1	30.0	257.50	538.707	78	78
Temperature 2	32.9	259.37	408.599	72	72
Oxygen 1	30.0	253.33	76.079	72	71
Oxygen 2	30.0	258.98	119.248	72	70

To determine moisture content, three samples weighing 150–200 g were dried to a constant weight at 105 °C and then gravimetrically analysed.

To determine volatile matter content, three 500 g sub-samples were pre-dried at room temperature for 48 hours and then ground to a 1 mm grain size using a Retsch mill. The percentage of volatile matter in the resulting sample was determined. Three 1 g test samples obtained after annealing the material in an oven (850 °C, 7 minutes, in a closed porcelain crucible) were gravimetrically analysed.

To determine the calorific value using the heat of combustion, three 1 g samples were analysed using an IKA Werke C2000 unit according to the technical norm ČSN EN 15 400 Solid recovered fuels – Determination of calorific value.

Energy balance

The energy balance of the process was calculated as follows:

$$E = (H_{after} \cdot m_{after}) - (H_{before} \cdot m_{before}) - P \cdot t$$

where E – energy balance, H_{after} – calorific value after drying, H_{before} – calorific value before drying, m_{waste} – mass of dried waste, P – air supply wattage (65 W), t – air supply function time.

Results and Discussion

Removal of water

Moisture contents were in accordance with the relative air humidity in the upper and lower layers in both methods. The largest decrease in moisture content was recorded when temperature was controlled in run No. 2, when in the lower layer it decreased by 27.54%. This is in accordance with Adani et al. (2002) and Sugni et al. (2005). The moisture content in the upper layer varied by about 10% from that recorded in the lower layer.

When the oxygen concentration was controlled the drying efficiency was the lowest. In fact, we even recorded a rise in moisture content in the upper layer. This was a result of water condensing on the ceiling of the reactor. Condensation occurred in all runs due to the low level of aeration, but in this run it was particularly marked (Table 3). These results are consistent with those of De Guardia et al. (2012).

Calorific value

The biggest change in the calorific value was recorded in run No. 1, in which temperature was controlled, which was because it was well aerated, little decomposition of organic matter and the low relative air humidity in the

Table 3 Values of moisture before and after the runs.

Run	Moisture before (%)	Moisture after in upper layer (%)	Moisture after in lower layer (%)	Desiccation of upper layer (%)	Desiccation of lower layer (%)
Temperature 1	41.80 ± 1.88	23.87 ± 3.20	13.74 ± 2.79	17.41	27.54
Temperature 2	43.64 ± 1.16	25.08 ± 0.65	22.87 ± 1.60	18.56	20.77
Oxygen 1	36.88 ± 1.61	37.82 ± 2.12	21.76 ± 0.65	-0.94	15.12
Oxygen 2	38.52 ± 1.26	15.81 ± 0.43	16.09 ± 0.80	22.71	22.43

$\alpha = 95\%$

Table 4 Calorific value before and after the process.

Run	LCV before (MJ/t)	LCV after in upper layer (MJ/t)	LCV after in lower layer (MJ/t)	LCV change in upper layer (%)	LCV change in lower layer (%)
Temperature 1	9,077.12 ± 73.78	13,818.18 ± 344.23	15,415.33 ± 253.22	34.31	41.12
Temperature 2	9,835.66 ± 21.21	13,610.96 ± 772.26	14,817.38 ± 668.71	27.74	33.62
Oxygen 1	12,358.60 ± 219.74	11,314.05 ± 51.51	14,548.64 ± 95.82	-9.23	15.05
Oxygen 2	11,102.23 ± 123.71	14,982.44 ± 197.09	17,001.06 ± 180.99	25.90	34.70

LCV – Lower calorific value, $\alpha = 95\%$

Table 5 Energetic balance of the process.

Run	Air supply operational time (s)	Energy consumed (MJ)	LCV average before drying (MJ/t)	LCV average after drying (MJ/t)	Energetic balance (MJ)
Temperature 1	439,338	28.12	9,077	14,617	138.07
Temperature 2	342,441	21.92	9,836	14,214	122.14
Oxygen 1	64,231	4.11	12,359	12,931	13.07
Oxygen 2	101,854	6.52	11,102	15,992	140.17

LCV – Lower calorific value

laboratory. For details, see Table 5 and Fig. 2. Similar results are reported by Adani et al. (2002) and Sugni et al. (2005).

In run No. 1, in which the concentration of oxygen was controlled, we recorded the smallest change in the calorific value and it even a decrease in the upper layer. This was, as mentioned previously, due to the significant condensation on the ceiling of the reactor. Similarly, a minor change in calorific value was recorded in the lower layer. As the rate of aeration was very low its drying effect was low.

Energetic balance

The best energetic balance was recorded in run No. 2, in which the oxygen concentration was controlled. This was due to a high calorific value and a significantly lower energy consumption than in the other runs. A very similar energetic balance was recorded when the temperature was controlled in run No. 1. When the temperature was controlled in run No. 2, the energetic balance was slightly worse than in the previously mentioned runs since the drying efficiency was lower and energy consumption higher. The worst energetic balance was recorded when the concentration of oxygen was controlled in run No. 1, due to a very high level of condensation (Table 4).

Both of the methods resulted in a positive yielded of energy. This means that even though some energy was used for drying, when burned, the dry waste produced more energy than when untreated waste was incinerated. Similar results were obtained by Cimpan and Wenzel (2013) and Economopoulos (2010).

Effect of ambient moisture

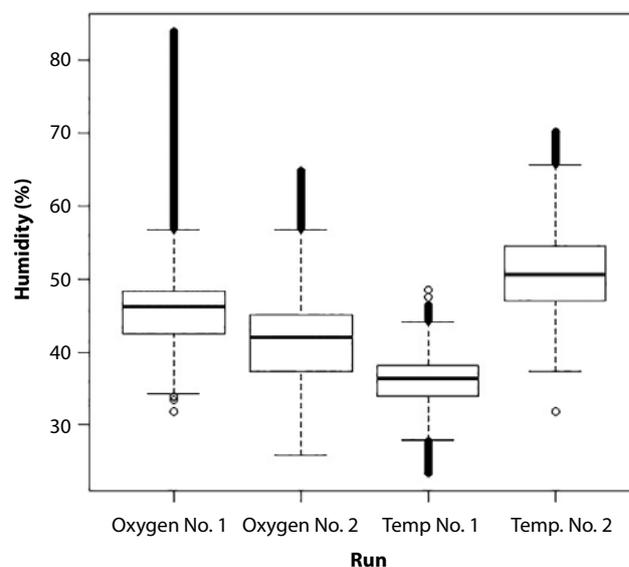
In the method in which temperature was controlled, the drying efficiency recorded in run No. 1 and run No. 2 differed. In run No. 1, the moisture decreased from $41.28 \pm 1.88\%$ to $23.87 \pm 3.20\%$ in the upper layer and to $13.74 \pm 2.79\%$ in the lower layer. In run No. 2, the moisture decreased from $43.64 \pm 1.16\%$ to $25.08 \pm 0.65\%$ in the upper layer and to $22.87 \pm 1.60\%$ in lower layer. The difference in drying efficiency was due to the difference in ambient air humidity (see Fig. 2). Apart from our study, the effect of ambient air humidity is mentioned only in one of the other studies on bio drying. Colomer-Mendoza et al. (2012) state that ambient air humidity, partic-

ularly the humidity of the air entering the reactor determines the efficiency of drying.

Spatial differences in temperature, moisture level and calorific value

The recordings depicted in Figs 3, 4, 5 and 6 reveal a big difference in temperature in the lower and upper layer in the reactor. This is well described by Adani et al. (2002) and Sugni et al. (2005) and is attributed to the lower part of the reactor being cooled by the surrounding air. This difference was recorded in all of the runs.

The literature indicates there is also a difference in the moisture content. This is due to differences in the drying efficiency in the lower and upper layers of the reactor (Table 3) as previously reported by Adani et al. (2002), Sugni et al. (2005) and Zhao et al. (2010). The difference can be explained by the fact that the moisture in the upper layer of the reactor is not removed as efficiently by the air supplied from the surroundings as it is in the lower layer. The recordings in Figs 3, 5 and 6 support this explanation. In general, there was a lower relative humidity in the lower layer than in the upper layer of the reactor. An exception is shown in Fig. 4, (method in which the concentration of oxygen was controlled, run No. 2), when the relative

**Fig. 2** Ambient air humidity recorded in the different runs.

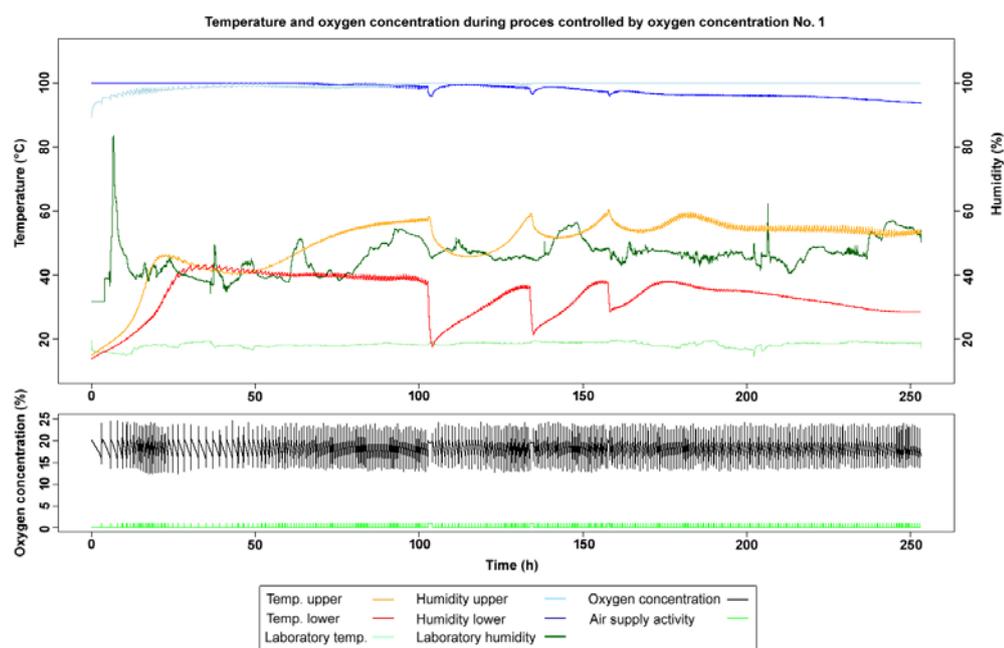


Fig. 3 Temporal dependence of the temperature, relative air humidity and oxygen level measured in the reactor and/or in the lab, temporal dependence of the air supply activity – method in which the concentration of oxygen was controlled, run No. 1.

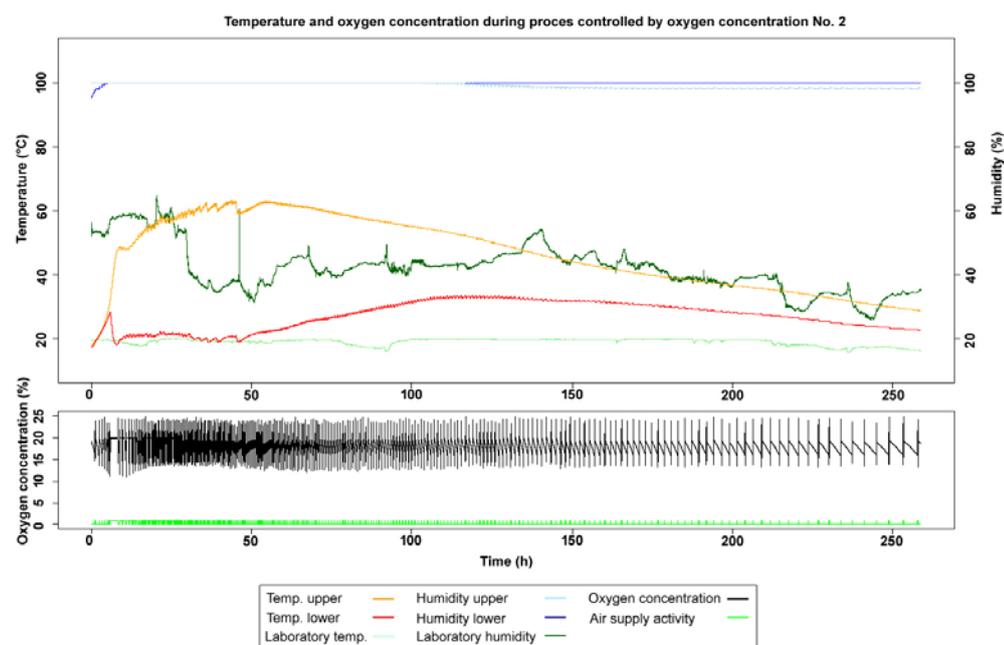


Fig. 4 Temporal dependence of the temperature, relative air humidity and oxygen level measured in the reactor and/or in the lab, temporal dependence of the air supply activity – method in which the concentration of oxygen was controlled, run No. 2.

Table 6 Changes in volatile matter content.

Run	Initial volatile matter content (%)	Final volatile matter content in upper layer (%)	Final volatile matter content in lower layer (%)	Change in vol. matt. in upper layer (%)	Change in vol. matt. in lower layer (%)
Temperature 1	63.24 ± 0.80	65.04*	65.44 ± 1.13	1.80	2.20
Temperature 2	69.04 ± 0.45	68.31 ± 0.42	68.25 ± 0.37	-0.73	-0.79
Oxygen 1	63.85 ± 0.42	70.73 ± 0.13	69.75 ± 0.44	6.88	5.90
Oxygen 2	68.64 ± 1.06	66.32 ± 0.80	70.78 ± 0.90	-2.32	2.14

* based on only one measurement, $\alpha = 95\%$

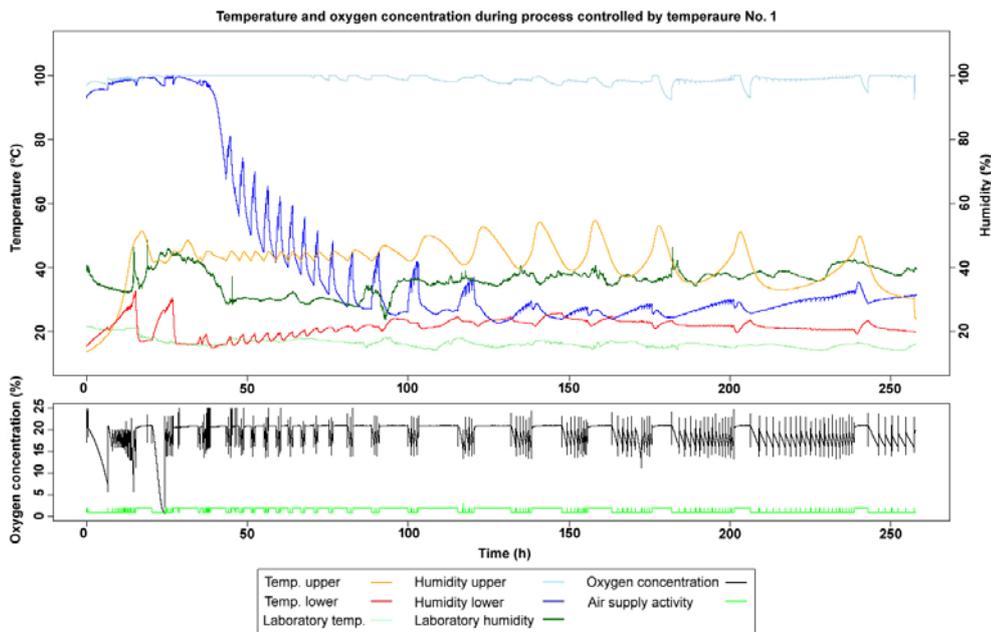


Fig. 5 Temporal dependence of the temperature, relative air humidity and oxygen level measured in the reactor and/or in the lab, temporal dependence of the air supply activity - method in which the temperature was controlled, run No. 1.

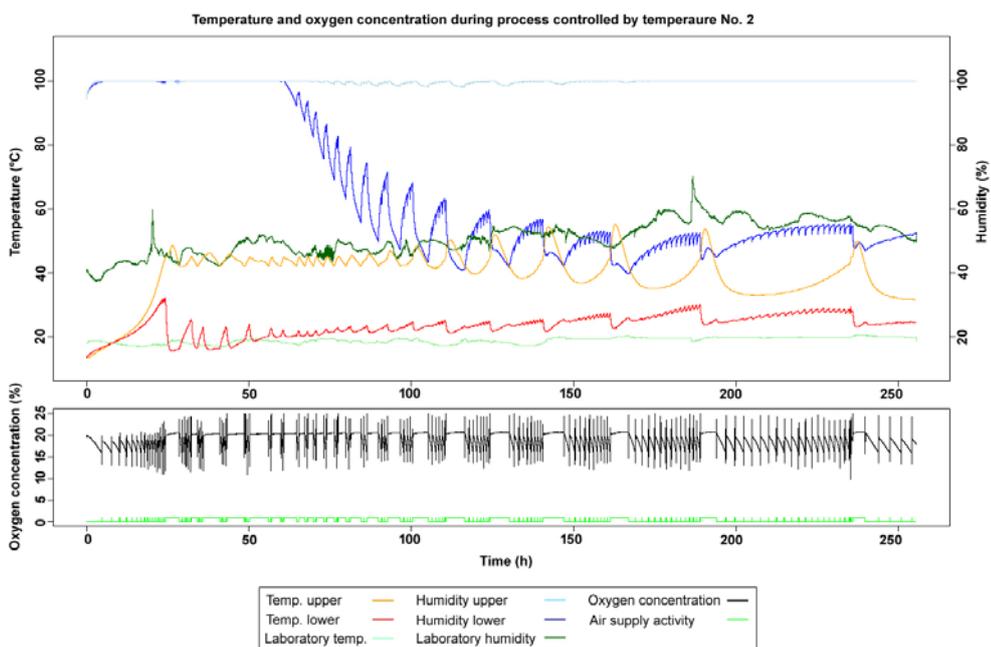


Fig. 6 Temporal dependence of the temperature, relative air humidity and oxygen level measured in the reactor and/or in the lab, temporal dependence of the air supply activity - method in which the temperature was controlled, run No. 2.

humidity recorded in the lower and upper layers are approximately comparable in the middle of the run. Subsequently, the relative humidity in the upper layer began to decrease. Relatively balanced moisture values recorded in Table 2 agree with Fig. 4. This effect may be due to a higher temperature being maintained for longer in the upper layer, which is indicated by the high average temperature and high cumulative temperature recorded in the upper layer. On that account, there was a large difference in cumulative temperature between the lower and the upper

layer (Tables 6 and 7). A low temperature in the lower layer and a great difference in temperature between the layers resulted in a reduced moistening of the air flowing through the lower layer. In the upper layer, where the temperature was higher, the air had a greater capacity for absorbing moisture and thus greater desiccation of the upper layer. The high temperature in the upper layer was a result of a high rate of decomposition of organic matter, possibly due to the high organic matter content of dry waste and an optimal concentration of oxygen. The high

Table 6 Average temperatures.

Run	Average temperature of upper layer (°C)	Average temperature of lower layer (°C)	Average difference in temperature (°C)
Temperature No. 1	41.30	21.28	20.02
Temperature No. 2	39.46	23.40	16.06
Oxygen No. 1	49.69	33.51	16.18
Oxygen No. 2	46.43	27.00	19.44

Table 7 Cumulative temperatures.

Run	Cumulative temperature of upper layer (°C·s)	Cumulative temperature of lower layer (°C·s)	Difference in Cumulative temperatures (°C·s)
Temperature No. 1	7 674 846	3 955 244	3 719 602
Temperature No. 2	7 267 152	4 309 049	2 958 103
Oxygen No. 1	9 063 866	6 112 895	2 950 971
Oxygen No. 2	8 657 955	5 033 945	3 624 011

level of decomposition was confirmed by the low value for volatile organic matter in the upper layer in the method in which the concentration of oxygen was controlled, run No. 2 (Table 5). Zawadzka et al. (2010) describe a similar phenomenon.

The differences in the values of the moisture and calorific content are similar in both runs of the method in which the temperature was controlled and run No. 1 in which the oxygen concentration was controlled. In which, the lower the moisture value recorded the higher the calorific value. The exception was the method in which the concentration of oxygen was controlled in run No. 2. Although the moisture content in the lower and upper layers was the same, there was a difference in the calorific values. This may be due to the decrease in organic matter content that resulted from the high rate of decomposition (Table 5).

Conclusions

Two methods of aeration were compared: in one the oxygen concentration was controlled and in the other temperature was controlled in the upper part of the reactor. A better performance in terms of removal of water and increase in calorific value was achieved using the method in which temperature was controlled. The net energetic balance of the bio drying process was positive for both methods. The method, in which the concentration of oxygen was controlled, was slightly better in terms of the energetic balance. The product, however, is much

more dependent on the composition of the dried waste used. Thus, it is suitable only for wastes of known composition.

Differences in temperature, moisture and calorific values recorded during the runs were similar to those reported in the literature. The only exception was the method, in which the concentration of oxygen was controlled, run No. 2. In this case, the opposite was recorded. In the upper layer, there was a lower moisture content and lower relative humidity than in the lower layer. This was probably because of the higher organic matter content of the sample processed. In combination with an optimal concentration of oxygen, the higher organic matter content resulted in a higher rate of decomposition and thus a higher temperature in the upper layer over a longer period of time. This brings us to the important finding that higher organic matter content and aeration regime settings invert the moisture gradient. In the future, this finding could be used to produce homogenous fuel mixtures.

There were differences in the drying efficiency recorded in run No. 1 and run No. 2 in which temperature was controlled. These differences were probably due to a difference in ambient air humidity. The characteristics of the ambient air (mainly humidity) could thus have an important effect on the drying efficiency. Ambient air characteristics are not considered in most studies on the bio drying of MSW.

Acknowledgements

The research was supported by the Technology Agency of the Czech Republic, project No. SS02030008, programme: Prostředí pro život.

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A REVIEW OF GLOBAL LAND COVER MAPS IN TERMS OF THEIR POTENTIAL USE FOR HABITAT SUITABILITY MODELLING

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ABSTRACT

Recently, there has been a significant increase in number of land cover maps available to researchers and they are now more commonly used. The broad variety requires some system for determining the differences between maps and for estimating their applicability for specific research purposes. We focused on comparing land cover maps from the point of view of how the land cover categories used characterize potentially suitable habitats for species. This comparison includes only freely available global land cover projects with resolutions from 1 km to 10 m. The criteria chosen were temporal and spatial resolution, number of classes and map precision. To demonstrate the differences, two areas of different sizes were always chosen. Our results reveal that maps can significantly vary in their estimates of different types of land cover, even at the same spatial resolution. Results also revealed that one type of vegetation in this area is poorly recorded in all land cover maps. Copernicus CGLS-LC100 and ESA CCI-LC maps appear to be the most suitable for evaluating potentially suitable habitats.

Keywords: comparison; Copernicus; ESA; FAO; habitat connectivity; habitat suitability

Introduction

The number of land cover maps and their use by researchers is greatly increasing. Earth observation satellites, such as Sentinel missions, Landsat missions, Terra and others provide data for these maps. Researchers can gain access to satellite imagery data with various levels of processing, such as an image of the Earth's surface with several bands, or an already processed map that is ready for analysis. Acquiring unprocessed satellite images could be a better option for a relatively small area, especially if it consists of a few images. Land cover maps can be created from satellite images in free open-source programs (Manton et al. 2005; Ndegwa Mundia and Murayama 2009; Barik et al. 2021). Precision of such maps depends not only on the quality of satellite imagery (Manton et al. 2005), but also on the classification approach (Li et al. 2017) and on type of data used (Novillo et al. 2018). However, for a global scale study, creating land cover maps from individual satellite images is extremely time consuming and this process will often exceed storage memory and processing capacities of a personal computer. Proper measuring of the precision of the final world map is an impossible task for an ordinary researcher. An average researcher, who is not specialized in processing remote sensing data, would be unable to correctly measure the precision of the final world map. Therefore, only land cover maps that do not require further processing were included in this comparative study. Some of these maps were created using data from several satellite sensors and replicating this method on a personal computer is unrealistic.

Land cover maps can be used in various fields of study: habitat connectivity (Ciudad et al. 2021), effect of changes in land use (Barik et al. 2021), conservation planning

(Falcucci et al. 2007), climate change (McMenamin et al. 2008) and forest monitoring (Rawat and Kumar 2015). Here we consider using land cover maps for estimating land cover of habitats that are potentially suitable for species on a global scale. For example, for epiphytic orchids, tropical forest is one of its suitable habitats. Evaluation of habitat suitability should be based on multiple sources of information (Manton et al. 2005; Hirzel and le Lay 2008). Nevertheless, land cover maps can be used to estimate, e.g., habitat connectivity (Cisneros-Araujo et al. 2021). Here we concentrate on usefulness of such maps for estimating site connectivity (e.g. that of an island or protected area) and habitat suitability for a species.

For the purpose of this study, maps should have certain characteristics. In order to correctly determine potentially suitable habitat, the map should temporally match that at the time the occurrence records were collected. It is worth considering using a series of land cover maps to determine changes over time, such as population dynamics (Ndegwa Mundia and Murayama 2009), the effect of changes in land cover on a species' habitat (Breininger et al. 2006), etc. Obtaining a time series of maps for a single project might minimize the bias in the resulting estimates of potentially suitable habitat, particularly if the maps were developed using the same algorithms and data from the same satellites. The map should have an appropriate spatial resolution. Choosing the appropriate spatial resolution is crucial for evaluating the pattern of potentially suitable habitat, since the final estimate of the extent of suitable habitat is highly dependent on the resolution of the land cover map (Rondinini et al. 2011; Ciudad et al. 2021). This decision is primarily based on the ecology of the species studied and purpose of the research. Overall, a too coarse resolution can omit habitat

fragmentation, resulting in an inaccurate estimate of the extent of suitable habitat. Displaying interactive maps in online map viewer, for example on the FAO GeoNetwork, should be treated with caution, as it tends to inflate the real resolution of the map. Number of classes needed for evaluating suitable habitat also depends on the aim of the study. A map with fewer classes, but higher resolution, would better reflect the average distribution of these classes than a coarser map with the same classification. However, for types of vegetation (forests, shrubs, etc.) it is important to provide multiple subcategories, since such categories have critical details for evaluating habitat suitability. E.g., category “Mosaic tree and shrub (> 50%) / herbaceous cover (< 50%)” in CCI-LC map (ESA 2017) provides more information than “Shrub Covered Areas” in GLC-SHARE map (Latham et al. 2014).

Materials and Methods

Maps were obtained from a variety of sources, the NASA Earthdata Search, ArcGIS, Zenodo and FAO GeoNetwork. 254 maps were found using NASA Earthdata Search using the keywords “Land use/Land cover” and “Global.” The majority were individual satellite images or specific small areas, which often included other categories of maps besides land cover maps (normalized difference vegetation index (NDVI), snow-free albedo, etc.). The names of projects and platforms (satellites or sensors) used in the development of global cover maps were the main results of this search. The agencies that provide final land cover maps, such as ESA, Copernicus and FAO, were discovered after searching individual projects and sensor names. GLOBCOVER and GLC-SHARE maps were found in the FAO GeoNetwork. The ArcGIS online Map Viewer was used to visually inspect some of the projects (MODIS, ESA). Maps GLC_FCS30, FROM-GLC10 were downloaded from Zenodo service. ESA and Copernicus maps were obtained directly from the respective agencies.

The final selection of land cover maps is listed in Table 1. All the maps listed are free to download and links to websites are given at the end of this paper. The maps were selected using the criteria outlined below. Two different sized areas were chosen to illustrate the variations in detail captured by each map. The large area is 60 × 115 km in size. It includes part of Croatia and nearby small islands: Čiovo (28.8 km²), Drvenik veli (12.07 km²), Drvenik mali (3.43 km²), Šolta (58.98 km²), Brač (396 km²), Hvar (297.4 km²), Paklinski islands (7.165 km²), Šćedro (8.36 km²). Small area is approximately 4 × 4 km in size. It includes a part of Čiovo island, which has a mosaic of forests, shrubs and open areas, and quite a large amount of urban areas along the coast and there does not seem to be any crops, only orchards, but definitely open herbaceous vegetation. Such a small area was chosen to demonstrate the accuracy of each map. For the smaller area, map classification was compared with Google Earth satellite imagery (Gorelick et al. 2017).

The following criteria for selecting maps were determined based on their potential use in evaluating habitat connectivity or habitat suitability:

Matching time periods

Some projects include land cover maps for earlier periods (ESA CCI-LC map is available from the year 1992), while others are for a specific short period of time (ESA GLOBCOVER map). There are projects that predict historical land use changes, such as ISLSCP II data collection (International Satellite Land-Surface Climatology Project, Initiative II), which includes 50 global time series data sets from 1986 to 1995 and describes historical changes in land use over a period of 300 years (1700–1990) (Goldewijk et al. 2007). Or, for example, a project from the United States Geological Survey (USGS), which is only for the years 1992–1993, but contains more data for this period of time. It provides not only a map of land cover, but also a global ecosystems map, simple biosphere model, biosphere–atmosphere transfer scheme, and vegeta-

Table 1 Final map selection.

Data provided by	Map name	Resolution	Number of classes	Approximate file size	Overall mapping accuracy	Temporal coverage
Copernicus	CGLS-LC100	100 m	23	53 GB	80.6% in 2015 80.3% in 2019	2015–2019
Food and Agriculture Organization of the United Nations (FAO)	GLC-SHARE	1000 m	11	35.2 MB	80.2%	1998–2012
European Space Agency (ESA)	CCI-LC	300 m	37	258 MB /2.3 GB	75.4%	1992 – present with one year delay
European Space Agency (ESA)	GLOBCOVER	300 m	22	374 MB	67.10%	12.2004 – 06.2006 01.2009 – 12.2009
Liangyun et al. 2020 (open access)	GLC_FCS30	30 m	9/16/24	21 GB	82.5%/71.4% /68.7%	2015, 2020
Gong et al. 2019 (open access)	FROM-GLC10	10 m	27 (2015) 11 (2017)	unknown	72.76%	2015, 2017

tion lifeform map. Monthly NDVI composites, however, are only available on the continent-scale map (GLCC).

Temporal resolution

The majority of the final products have the same temporal resolution (one year), which may be insufficient for analyses requiring a finer temporal resolution (Ciudad et al. 2021). Unfortunately, there is no global land cover map with a lower temporal resolution. Indexes, characterizing vegetation, such as normalized difference water index or normalized difference vegetation index are produced monthly and can be used as substitution for a land cover map (Teng et al. 2021). If a land cover map is needed for a particular time period, for a smaller area it can be created using satellite imagery (Pennington et al. 2008; Li et al. 2017).

Spatial resolution

Maps with resolutions greater than 1 km were excluded from the comparison due to the loss of landscape features (islands, water bodies, etc.) at such resolutions.

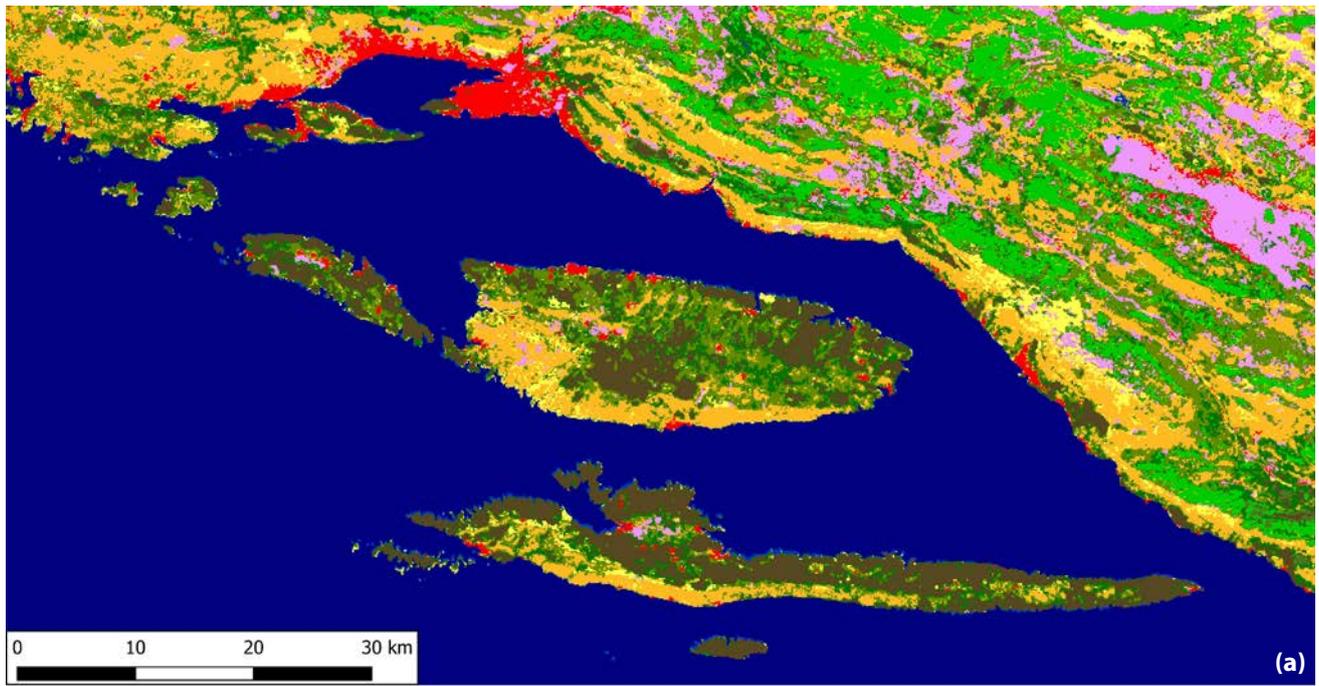
Highest global land cover map resolution that is currently available is 30 m (considering only publicly available projects from major agencies, for which this precision was thoroughly evaluated). There are companies that provide higher-resolution maps, for example, 10 m BaseVue maps from 2005 to the present time (MAXAR 2021). However, these maps have to be purchased and are provided on request, for a user-defined area. Alternatively, such maps can be obtained from open projects, such as FROM-GLC10 with a resolution of 10 m (Gong et al. 2019).

File size

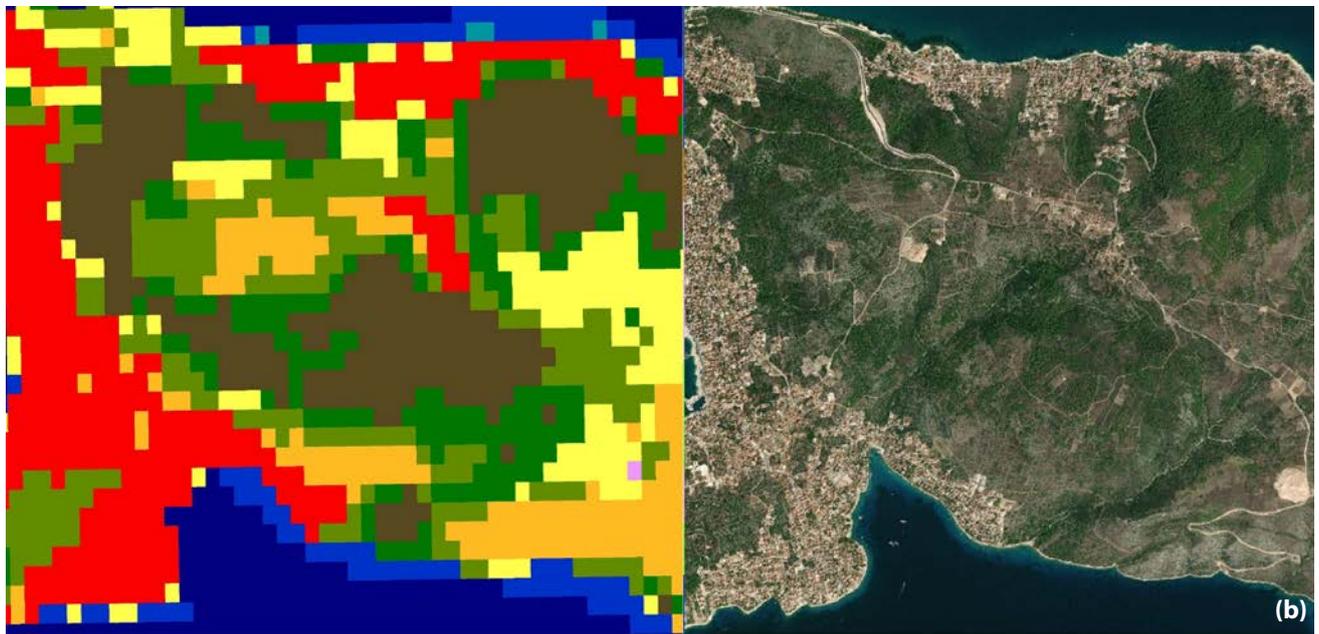
Since this is primarily determined by map resolution, a direct comparison would be unreasonable. Even so, due to the difference in raster compression methods, the size of the final product will vary between maps with a similar resolution. Distributors provide well-compressed maps, but it should be mentioned that file size can drastically change after re-saving, for example, after exporting categories into separate files. This may present a problem, es-

Table 2 Land cover classes included on CGLS-LC100 map, taken from Buchhorn et al. (2020). Corresponding land cover map is shown in Fig. 1.

Land cover class	Definition
Closed forest, evergreen needles	Tree canopy > 70%, almost all trees with needles remain green all year. Canopy is never without green foliage.
Closed forest, evergreen, broad leaf	Tree canopy > 70%, almost all trees are broadleaved and remain green all year. Canopy is never without green foliage.
Closed forest, deciduous needles	Tree canopy > 70%, almost all trees bear needles and are deciduous.
Closed forest, deciduous broad leaf	Tree canopy > 70%, almost all trees are broadleaved and deciduous.
Open forest, evergreen needles	Top layer: trees 15–70% and second layer: mixed of shrubs and grassland, almost all the trees bear needles and are evergreen. Canopy is never without green foliage.
Open forest, evergreen broad leaf	Top layer: trees 15–70% and second layer: mixed of shrubs and grassland, almost all the trees are broadleaved and evergreen. Canopy is never without green foliage.
Open forest, deciduous needles	Top layer: trees 15–70% and second layer: mixed of shrubs and grassland, all the trees bear needles and are deciduous.
Open forest, deciduous broad leaf	Top layer: trees 15–70% and second layer: mixed of shrubs and grassland, all the trees are broadleaved and deciduous.
Shrubs	These are woody perennial plants with persistent woody stems and no main stem and are less than 5 m tall the foliage of which can be either evergreen or deciduous
Herbaceous vegetation	Plants without persistent stem or shoots above ground and lacking definite firm structure. Tree and shrub cover is less than 10%.
Cultivated and managed vegetation/agriculture (cropland)	Land covered with temporary crops followed by harvest and a bare soil (e.g., single and multiple cropping systems). Note that perennial woody crops will be classified as the appropriate type of forest or shrub land cover.
Urban / built up	Land covered by buildings and other man-made structures.
Bare / sparse vegetation	Lands with exposed soil, sand, or rocks with never more than 10% covered with vegetation.
Snow and ice	Lands under snow or ice throughout the year.
Permanent water bodies	Lakes, reservoirs, and rivers. Can be either fresh or salt-water.
Temporary water bodies	
Herbaceous wetland	Lands with a permanent mixture of water and herbaceous or woody vegetation. The vegetation can be present in either salt, brackish, or fresh water.
Open sea	Oceans, seas. Can be either fresh or salt-water bodies.



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|----------------------------------------------------------|--------------------------------------|------------------------------------|
| 0 | Permanent water bodies | Open forest, evergreen needle leaf |
| Shrubs | Herbaceous wetland | Open forest, deciduous needle leaf |
| Herbaceous vegetation | Moss and lichen | Open forest, deciduous broad leaf |
| Cultivated and managed vegetation/agriculture (cropland) | Closed forest, evergreen needle leaf | Open forest, mixed |
| Urban / built up | Closed forest, deciduous broad leaf | Open forest, unknown |
| Bare / sparse vegetation | Closed forest, mixed | Open sea |
| Snow and Ice | Closed forest, unknown | |



0 1 2 km

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|----------------------------------------------------------|--------------------------------------|-----------------------------------|
| Shrubs | Urban / built up | Open forest, deciduous broad leaf |
| Herbaceous vegetation | Closed forest, evergreen needle leaf | Open forest, unknown |
| Cultivated and managed vegetation/agriculture (cropland) | Closed forest, unknown | |

Fig. 1 Land cover map CGLS-LC100, (a) less detailed, legend includes all map classes; (b) more detailed, only the map classes present in the area are shown.

pecially with maps with high spatial resolution and large compressed file sizes.

Number of classes

Some projects provide comparison of maps classification system with the Land Cover Classification System (LCCS). FAO created this method to provide a standardized structure for land cover classification and mapping. As a result, such projects are easier to compare.

Map precision

Due to the global scale of the data, the most reliable information on the precision of the final map can be obtained from the map's provider. The precision of a map may vary depending on the number of classes (for example, in the GLC FCS30 map), so the accuracy of same map depends on the number of classes used. Furthermore, the precision of a project's estimate of a specific land cover category can vary; this information is usually given in validation reports.

Results

Copernicus global land service: land cover 100 m (Buchhorn et al. 2020)

This project provides a comparison of map land cover classes with definition from LCCS classification system (Table 2). In addition to a map with 18 land cover classes, Copernicus provides layers that describe probability and quality of classification for each pixel. This map is also accompanied by cover layers, which define the percentage of pixels covered by a particular class pixel (forest, herbaceous vegetation, shrub, and bare soil) (Buchhorn et al. 2021).

Fig. 1a shows that the map not only depicts a high overall diversity of different types of land cover, but also a diversity on small islands and even the smallest island. Fig. 1b gives a clear indication of the level of detail recorded. There are clear similarities in the detail in satellite imagery and a map characterization of land cover, even the shapes of urban areas are accurate. The only inaccur-

Table 3 CCI-LC map categories, adapted from ESA (2017). Corresponding land cover map is shown in Fig. 2.

Global scale class	Regional scale class
Cropland, rainfed	Cropland, rainfed, herbaceous cover
	Cropland, rainfed, tree or shrub cover
Cropland, irrigated or post-flooding	
Mosaic cropland (> 50%) / natural vegetation (tree, shrub, herbaceous cover) (< 50%)	
Mosaic natural vegetation (tree, shrub, herbaceous cover) (> 50%) / cropland (< 50%)	
Tree cover, broadleaved, evergreen, closed to open (> 15%)	
Tree cover, broadleaved, deciduous, closed to open (> 15%)	Tree cover, broadleaved, deciduous, closed (> 40%)
	Tree cover, broadleaved, deciduous, open (15–40%)
Tree cover, coniferous, evergreen, closed to open (> 15%)	Tree cover, coniferous, evergreen, closed (> 40%)
	Tree cover, coniferous, evergreen, open (15–40%)
Tree cover, coniferous, deciduous, closed to open (> 15%)	Tree cover, coniferous, deciduous, closed (> 40%)
	Tree cover, coniferous, deciduous, open (15–40%)
Tree cover, mixed leaf type (broadleaved and coniferous)	
Mosaic tree and shrub (> 50%) / herbaceous cover (< 50%)	
Mosaic herbaceous cover (> 50%) / tree and shrub (< 50%)	
Shrubland	Evergreen shrubland
	Deciduous shrubland
Grassland	
Lichens and mosses	
Sparse vegetation (tree, shrub, herbaceous cover) (< 15%)	Sparse tree cover (< 15%)
	Sparse shrub cover (< 15%)
	Sparse herbaceous cover (< 15%)
Tree cover, flooded, fresh or brackish water	
Tree cover, flooded, saline water	
Shrub or herbaceous cover, flooded, fresh/saline/brackish water	
Urban areas	
Bare areas	Consolidated bare areas
	Unconsolidated bare areas
Water bodies	
Permanent snow and ice	

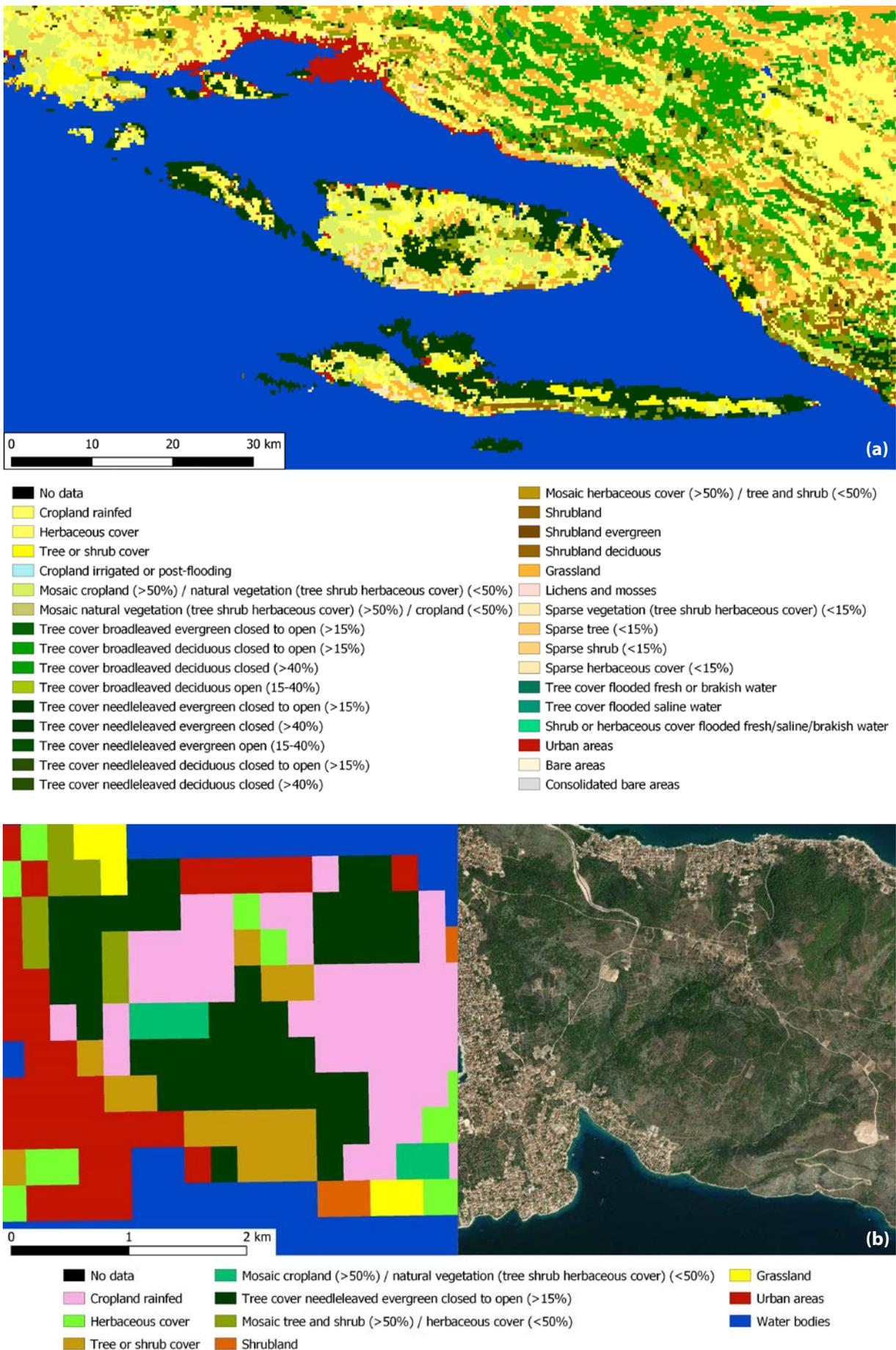


Fig. 2 Land cover map CCI-LC, (a) less detailed, legend includes all map classes; (b) more detailed, only map classes present in the area are shown.

racy is that bare soil is categorized as cropland (one pink pixel bottom right). However, due to the small size of this patch, this misclassification is negligible, as it indicates there is a difference in the landscape.

European Space Agency: Climate Change Initiative Land Cover map (CCI-LC) (ESA 2017)

CCI-LC map categories are compatible with the LCCS coding. This map has two scales of categories (Table 3), in some areas, where certain types cannot be defined, global scale classes are substituted for regional scale classes.

CCI-LC correctly indicates the uneven distribution of different types of land cover, as shown in Fig. 2a, and it also includes the smallest islands. Visual estimates of overall diversity is difficult due to the similar colouration of categories (in Fig. 2a categories have the colours in ESA). This is not a problem if the map has already been

downloaded. The disadvantage is that due to the higher spatial resolution, the majority of online map viewers will display this map in a very misleading manner (the map can be “upscaled” to a higher spatial resolution for a faster display). It is impossible to determine whether the area of interest is classified on a regional or global scale (all cropland and needle leaved forest classes are indistinguishable in terms of colour). In Fig. 2b, similar colours were changed. Fig. 2b reveals that the map provides significantly less detail than Copernicus, but overall it is similar. Orchards were identified as rain fed croplands, which are categorized as cropland in this map.

Food and agriculture organization of the United Nations: Global land cover SHARE (GLC-SHARE) (Latham et al. 2014)

This project’s map has only 11 classes, which are not compatible with the LCCS classification system, but in-

Table 4 Map categories with descriptions, taken from Latham et al. 2014.

Land Cover	Description
Artificial Surfaces	This category includes any type of area with a predominantly artificial surface. Any urban or related feature is included, for example urban parks (parks, parkland, sport facilities). It also includes industrial areas, waste dumps and extraction sites.
Cropland	Herbaceous crops: includes cultivated herbaceous plants (graminoids or herbaceous plants) and crops used for hay. All the non-perennial crops that do not last for more than two growing seasons and crops like sugar cane where the upper part of the plant is regularly harvested while the root system can remain for more than one year in the field are included.
	Woody crops: includes permanent crops (trees and/or shrub crops) and includes all types of orchards and plantations (fruit trees, coffee and tea plantation, oil palm, rubber plantation, Christmas trees etc.).
	Multiple or layered crops: includes different land cover situations: <ul style="list-style-type: none"> – Two layers of different crops (woody + herbaceous): A common case is the presence of woody crops (trees or shrubs) and herbaceous crops, such as, wheat fields with olive trees in the Mediterranean area and intense horticulture, oasis or typical coastal African agriculture where the cover for herbaceous fields is provided by palm trees, etc. – Presence of one important layer of natural vegetation (mainly trees) that cover one layer of cultivated crops: a typical example are coffee plantations shadowed by natural trees in the equatorial area of Africa.
Grassland	Includes any geographic area dominated by natural herbaceous plants (grasslands, prairies, steppes, and savannahs) with a cover of 10% or more, irrespective of different human and/or animal activities, such as: grazing, selective fire management etc. Woody plants (trees and/or shrubs) can be present providing their cover is less than 10%.
Tree covered areas	Includes any geographic area dominated by trees with a cover of 10% or more. Other types of plants (shrubs and/or herbaceous) can be present, even at a density greater than the trees. Areas planted with trees for afforestation purposes and forest plantations are included in this category, which also includes areas seasonally or permanently flooded with fresh water, but not coastal mangroves.
Shrubs covered areas	Includes any geographic area dominated by shrubs with a cover of 10% or more. Other types of plants (herbaceous) can be present, even at a density greater than shrubs.
Herbaceous vegetation, aquatic or regularly flooded	Includes any geographic area dominated by natural herbaceous vegetation (cover of 10% or more) that is permanently or regularly flooded by fresh or brackish water (swamps, marsh areas etc.). Flooding must persist for at least 2 months per year to be considered regular. Woody vegetation (trees and/or shrubs) can be present if their cover is less than 10%.
Mangroves	Includes any geographical area dominated by woody vegetation (trees or shrubs) with a cover of 10% or more that is permanently or regularly flooded by salt and/or brackish water located in coastal areas or in river deltas.
Sparse vegetation	Includes any geographic areas where the cover of natural vegetation is between 2% and 10%. This includes permanently or regularly flooded areas.
Bare soil	Includes any geographic area dominated by natural abiotic surfaces (bare soil, sand, rocks, etc.) where the natural vegetation is absent or almost absent (covers less than 2%) and areas regularly flooded by inland water (lake shores, river banks, salt flats etc.), but not coastal areas affected by the tidal movement of salt water.
Snow, glaciers	Includes any geographic area covered by snow or glaciers persistently for 10 months or more.
Waterbodies	Includes any geographic area covered for most of the year by inland water bodies. In some cases the water can be frozen for part of the year (less than 10 months). Because the geographic extent of water bodies can change, boundaries must be consistent with class 11 according to the dominant situation during a year and/or many years.

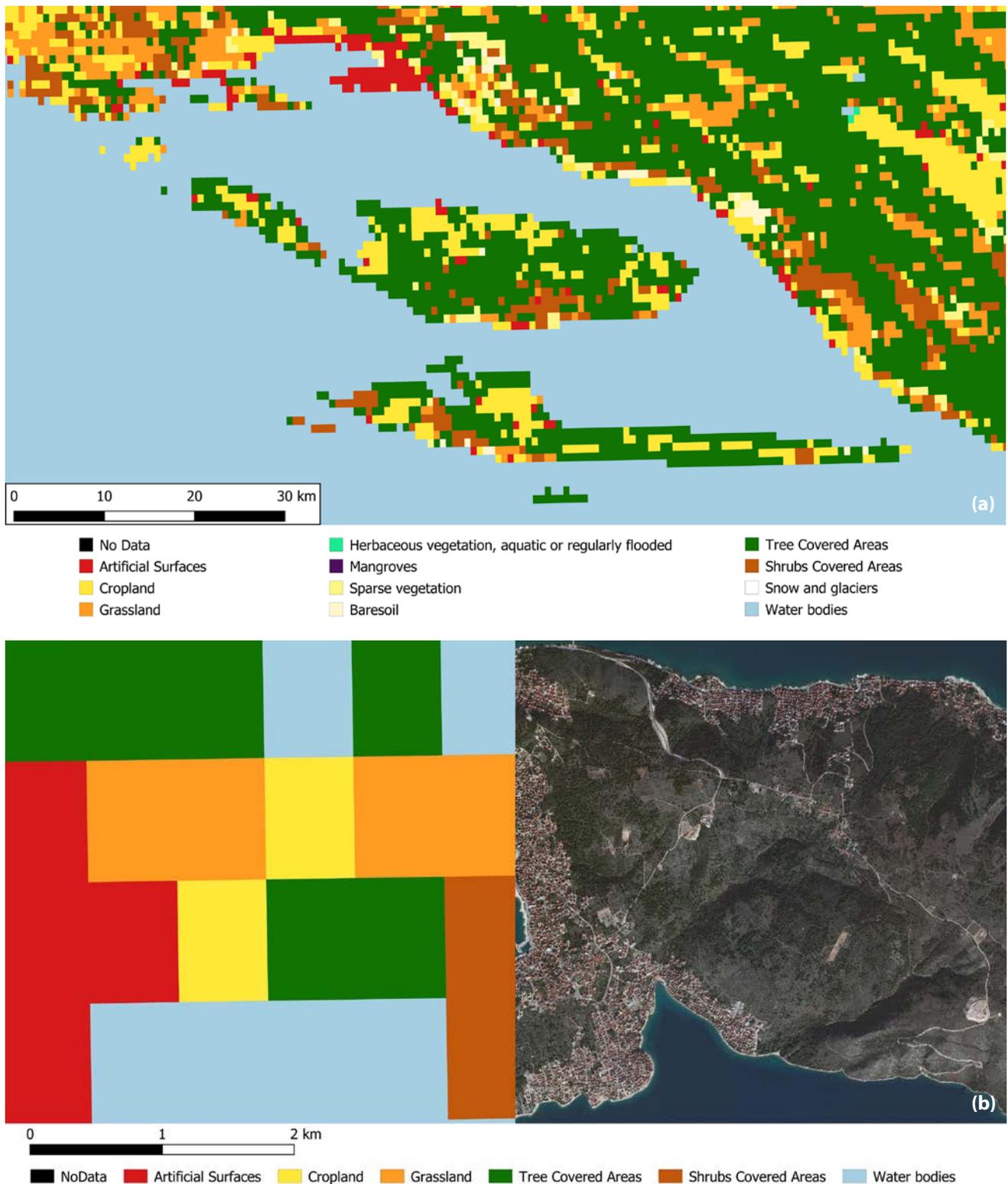


Fig. 3 Land cover map GLC-SHARE, (a) less detailed, legend includes all map classes; (b) more detailed, only map classes present in the area.

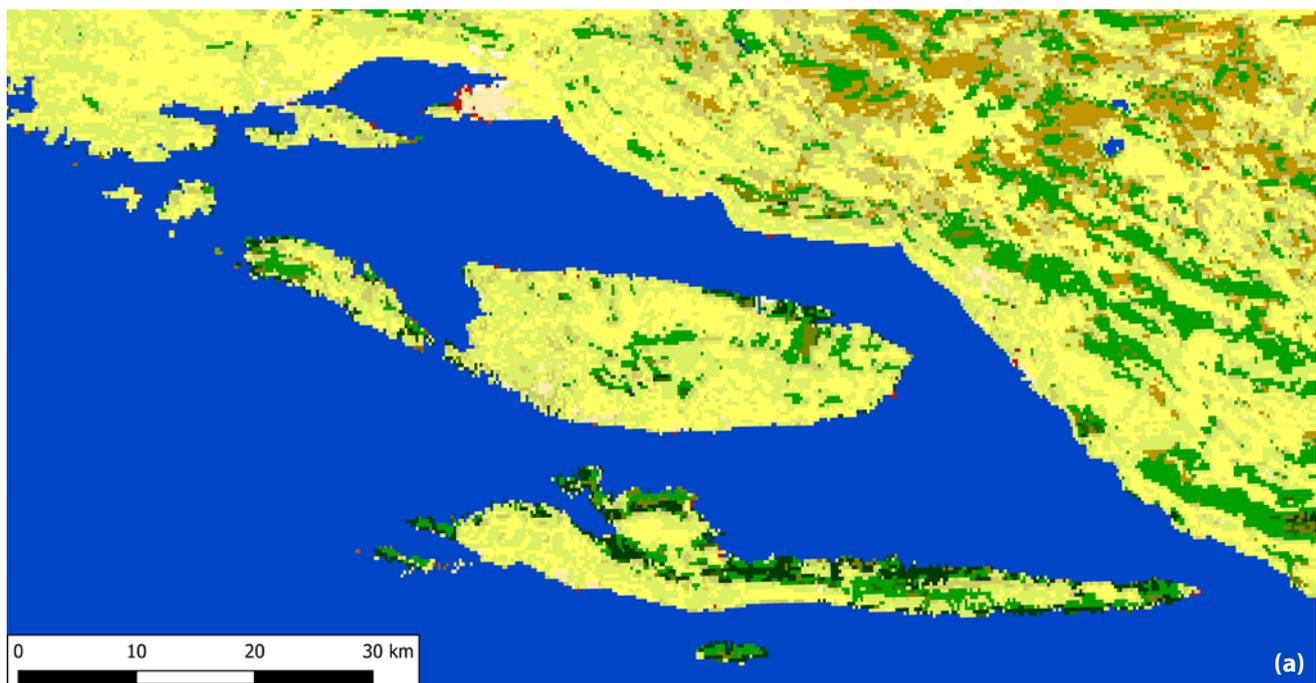
cludes a detailed description of each class (Table 4). This map not only doesn't accurately capture landscape features because of its resolution and number of classes, but it also omits several important details: distribution of urban areas, smaller islands, shape of coast line and so on (Fig. 3). Such resolution is hardly suitable for modelling species habitats.

European Space Agency: GlobCover (Arino et al. 2012)

Classification of GlobCover (Table 5) is compatible with the LCCS system. Fig. 4a shows that the map of this region does not truly represent urban areas (there should be more red colour along the coast, as in Fig. 2a, at a similar resolution). Fig. 4b reveals another misclassification: forest was not identified.

Table 5 Land cover classes of GLOBCOVER map, adapted from Arino et al. (2012).

Land cover classes	Land cover classes
Post-flooding or irrigated cropland (or aquatic)	Closed to open (> 15%) (broadleaved or coniferous, evergreen or deciduous) shrubland (< 5 m)
Rainfed cropland	Closed to open (> 15%) herbaceous vegetation (grassland, savannas or lichens/mosses)
Mosaic cropland (50–70%) / vegetation (grassland/shrubland/forest) (20–50%)	Sparse (< 15%) vegetation
Mosaic vegetation (grassland/shrubland/forest) (50–70%) / cropland (20–50%)	Closed to open (> 15%) broadleaved forest regularly flooded (semi-permanently or temporarily) – Fresh or brackish water
Closed to open (> 15%) broadleaved evergreen or semi-deciduous forest (> 5 m)	Closed (> 40%) broadleaved forest or shrubland permanently flooded – Saline or brackish water
Closed (> 40%) broadleaved deciduous forest (> 5 m)	Closed to open (> 15%) grassland or woody vegetation on regularly flooded or waterlogged soil – Fresh, brackish or saline water
Open (15–40%) broadleaved deciduous forest/woodland (> 5 m)	Artificial surfaces and associated areas (Urban areas > 50%)
Closed (> 40%) coniferous evergreen forest (> 5 m)	Bare areas
Open (15–40%) coniferous deciduous or evergreen forest (> 5 m)	Water bodies
Closed to open (> 15%) mixed broadleaved and coniferous forest (> 5 m)	Permanent snow and ice
Mosaic forest or shrubland (50–70%) / grassland (20–50%)	No data (burnt areas, clouds...)
Mosaic grassland (50–70%) / forest or shrubland (20–50%)	



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| <ul style="list-style-type: none"> ■ Post-flooding or irrigated croplands (or aquatic) ■ Rainfed croplands ■ Mosaic cropland (50-70%) / vegetation (grassland/shrubland/forest) (20-50%) ■ Mosaic vegetation (grassland/shrubland/forest) (50-70%) / cropland (20-50%) ■ Closed to open (>15%) broadleaved evergreen or semi-deciduous forest (>5m) ■ Closed (>40%) broadleaved deciduous forest (>5m) ■ Open (15-40%) broadleaved deciduous forest/woodland (>5m) ■ Closed (>40%) needleleaved evergreen forest (>5m) ■ Open (15-40%) needleleaved deciduous or evergreen forest (>5m) ■ Closed to open (>15%) mixed broadleaved and needleleaved forest (>5m) ■ Mosaic forest or shrubland (50-70%) / grassland (20-50%) ■ Mosaic grassland (50-70%) / forest or shrubland (20-50%) ■ Closed to open (>15%) shrubland (<5m) (broadleaved or needleleaved, evergreen or deciduous) | <ul style="list-style-type: none"> ■ Closed to open (>15%) herbaceous vegetation (grassland, savannas or lichens/mosses) ■ Sparse (<15%) vegetation ■ Closed to open (>15%) broadleaved forest regularly flooded (semi-permanently or temporarily) - Fresh or brackish water ■ Closed (>40%) broadleaved forest or shrubland permanently flooded Saline or brackish water ■ Closed to open (>15%) grassland or woody vegetation on regularly flooded or waterlogged soil - Fresh, brackish or saline water ■ Artificial surfaces and associated areas (Urban areas >50%) ■ Bare areas ■ Water bodies ■ Permanent snow and ice ■ No data (burnt areas, clouds,...) |
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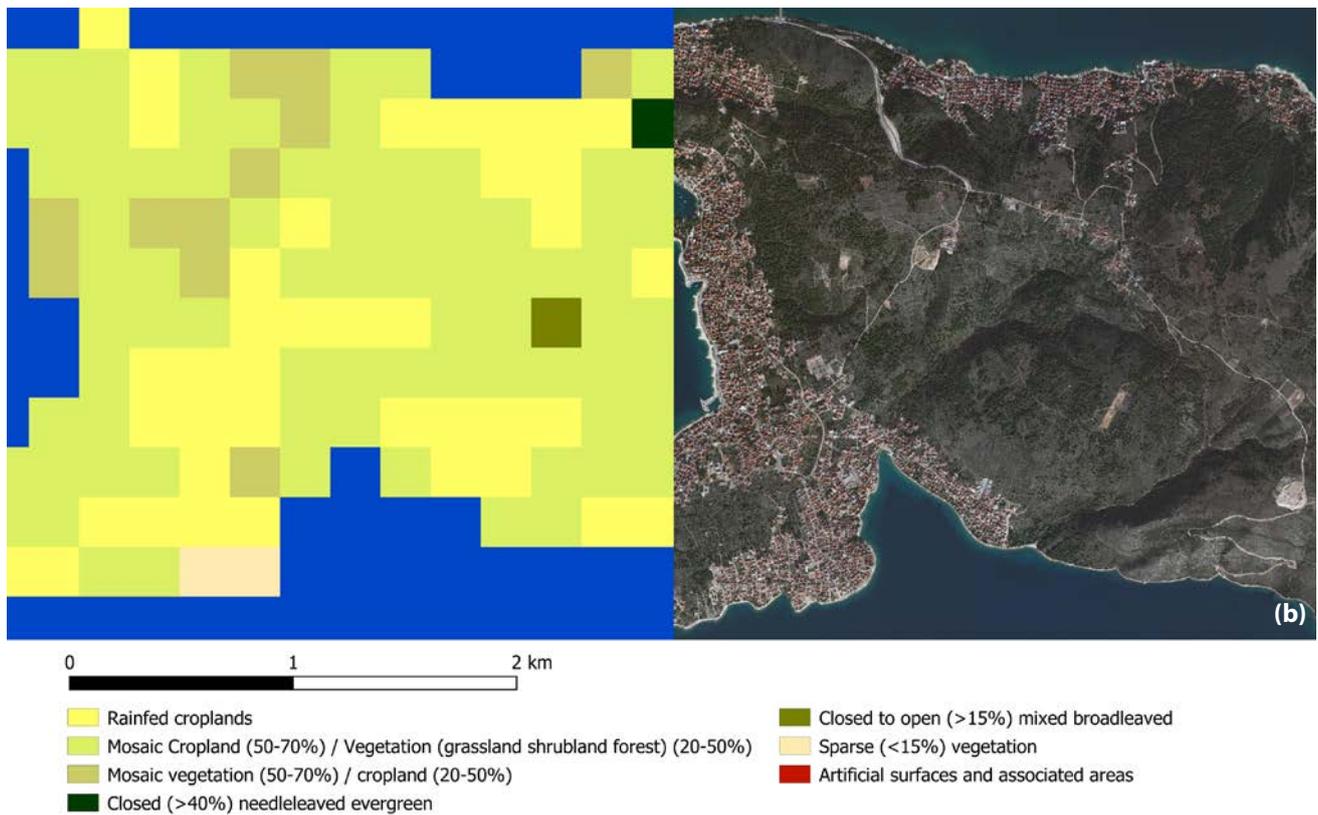


Fig. 4 Land cover map GlobCover: (a) less detailed, legend includes all map classes; (b) more detailed, only map classes present in the area.

GLC_FCS30 map, open access (Liangyun et al. 2020)

This map has 24 classes (Table 6), but is not compatible with the LCCS classification system. There are some misclassifications in this map along the coastline, but because of its resolution, it captures the finer details (Fig. 5). It also has some minor artefacts, such as: “orchards” along roads; small patches of “shrubland” within “closed forest”, despite the fact that in the various satellite images forest

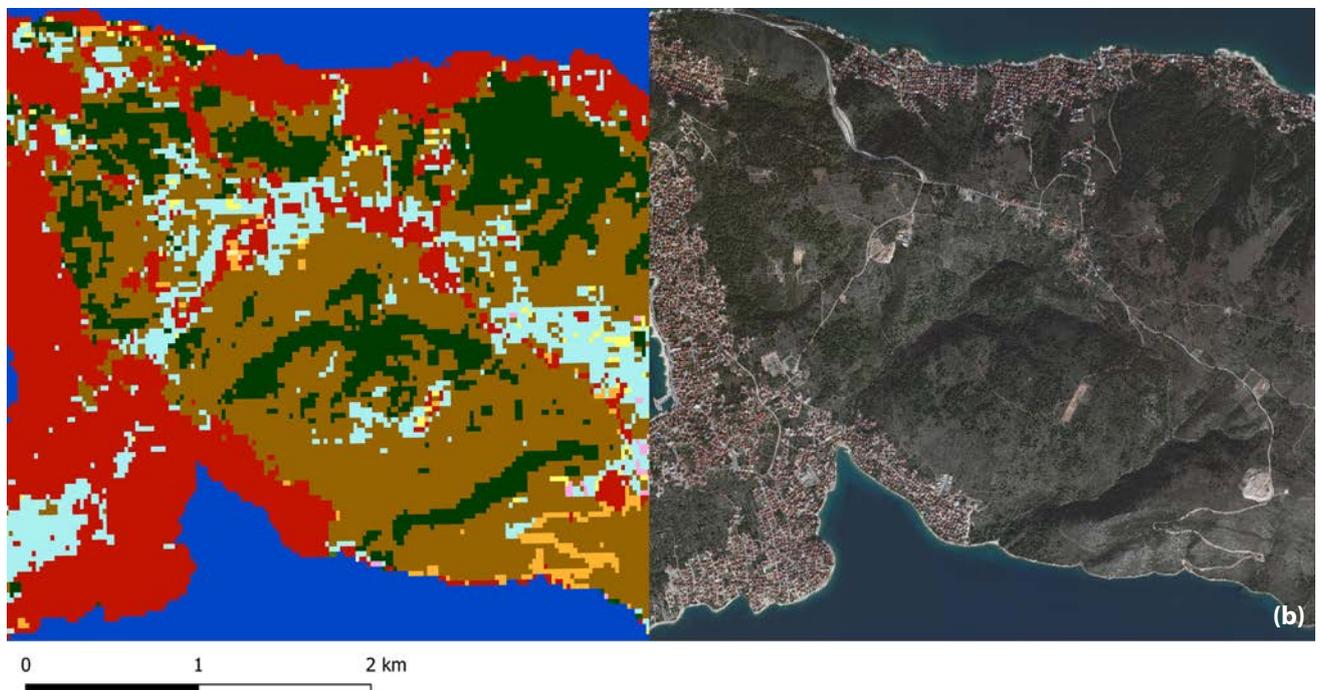
appears to be homogeneous at such locations; occasionally roads are classified as “herbaceous cover”. These minor misclassifications are correctable. However, this map has a more serious problem. Since sections of these maps overlap (Fig. 6a) and classification in the overlapping sections differs (Fig. 6b), these maps must be processed before they can be used for analysis. In contrast, the Copernicus (CGLS-LC100 map) map does not have such a

Table 6 Land cover classes GLC_FCS30 map, adapted from Liangyun et al. (2020). Classes with (*) are not present in map legend, these classes were added to categorize the legend.

Land cover classes		Land cover classes	
Cropland classes*	Rainfed cropland	Shrubland	Evergreen shrubland
	Orchard		Deciduous shrubland
	Irrigated cropland	Herbaceous cover	
Evergreen forest types*	Open evergreen broadleaved forest	Tree or shrub cover	
	Open evergreen broadleaved forest	Grassland	
	Open evergreen needle-leaved forest	Sparse vegetation	Sparse shrubland
	Closed evergreen needle-leaved forest		Sparse herbaceous vegetation
Deciduous forest types*	Open deciduous broadleaved forest	Lichens and mosses	
	Closed deciduous broadleaved forest	Wetlands	
	Closed deciduous needle-leaved forest	Impervious surfaces	
	Open deciduous needle-leaved forest	Bare areas	Unconsolidated bare areas
Mixed leaf forest types*	Open mixed broadleaved and needle-leaved forest		Consolidated bare areas
	Closed mixed broadleaved and needle-leaved forest	Water body	
		Permanent ice and snow	



- | | | |
|---------------------------------------------------------|-----------------------------------------------------------|------------------------|
| 0 | Open evergreen needle-leaved forest ($0.15 < fc < 0.4$) | Wetlands |
| Rainfed cropland | Closed evergreen needle-leaved forest ($fc > 0.4$) | Impervious surfaces |
| Herbaceous cover | Open mixed leaf forest (broadleaved and needle-leaved) | Bare areas |
| Tree or shrub cover (Orchard) | Closed mixed leaf forest (broadleaved and needle-leaved) | Water body |
| Irrigated cropland | Shrubland | Permanent ice and snow |
| Open deciduous broadleaved forest ($0.15 < fc < 0.4$) | Grassland | |
| Closed deciduous broadleaved forest ($fc > 0.4$) | Sparse vegetation ($fc < 0.15$) | |



- | | | |
|-------------------------------|------------------------------------------------------|---------------------|
| Rainfed cropland | Closed evergreen needle-leaved forest ($fc > 0.4$) | Impervious surfaces |
| Herbaceous cover | Shrubland | Water body |
| Tree or shrub cover (Orchard) | Grassland | |

Fig. 5 Land cover map GLC_FCS30, (a) less detailed, legend includes all map classes; (b) more detailed, only includes classes present in the area.

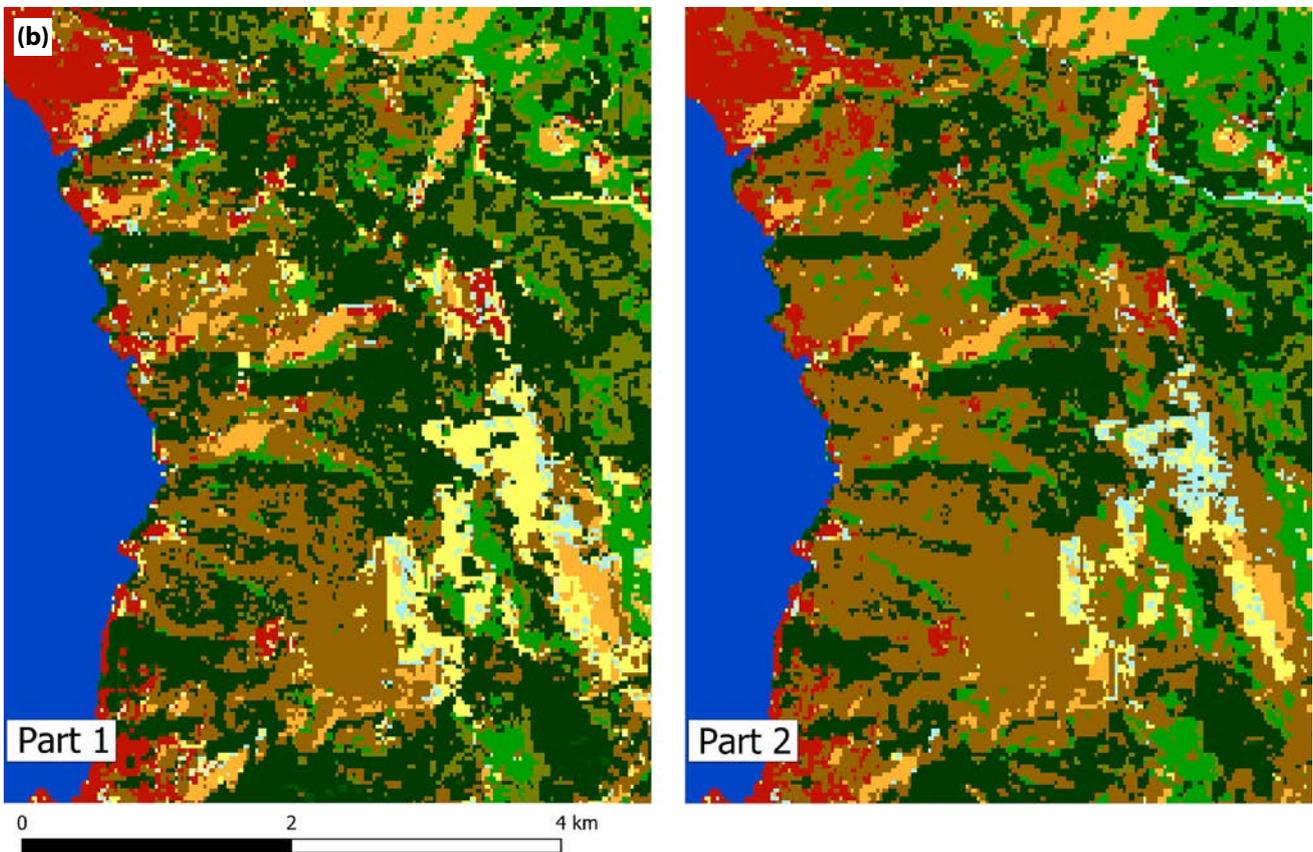
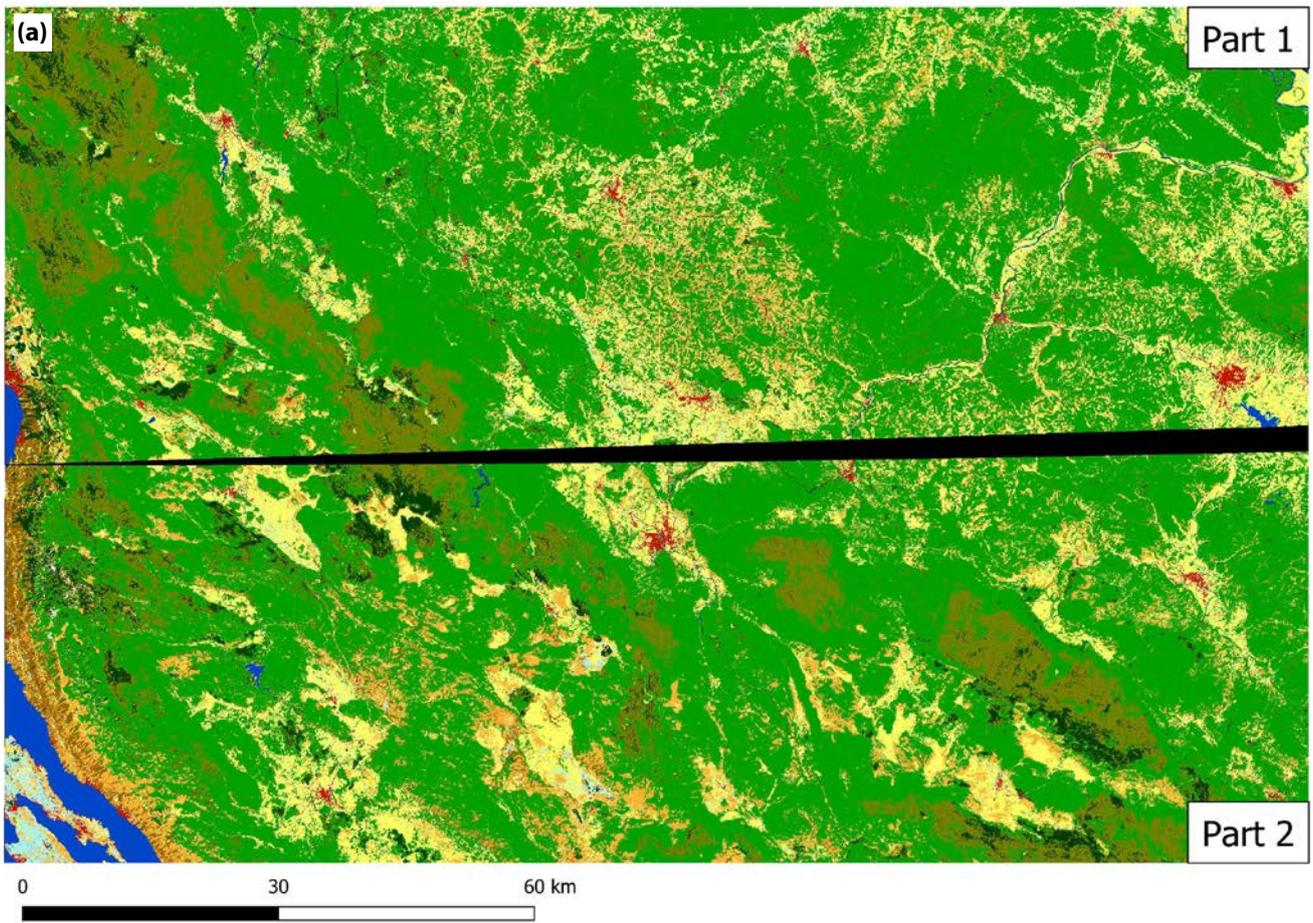


Fig. 6 Land cover map GLC_FCS30, (a) alignment between part 1 (E15N50) and part 2 (E15N45), extent of the area: 45.38, 14.9; 44.48, 16.82; (b) differences between overlapping sections, and the extent of the area: 44.98, 14.91, 44.93, 14.98. Parts' codes are section designations of GLC FCS30.

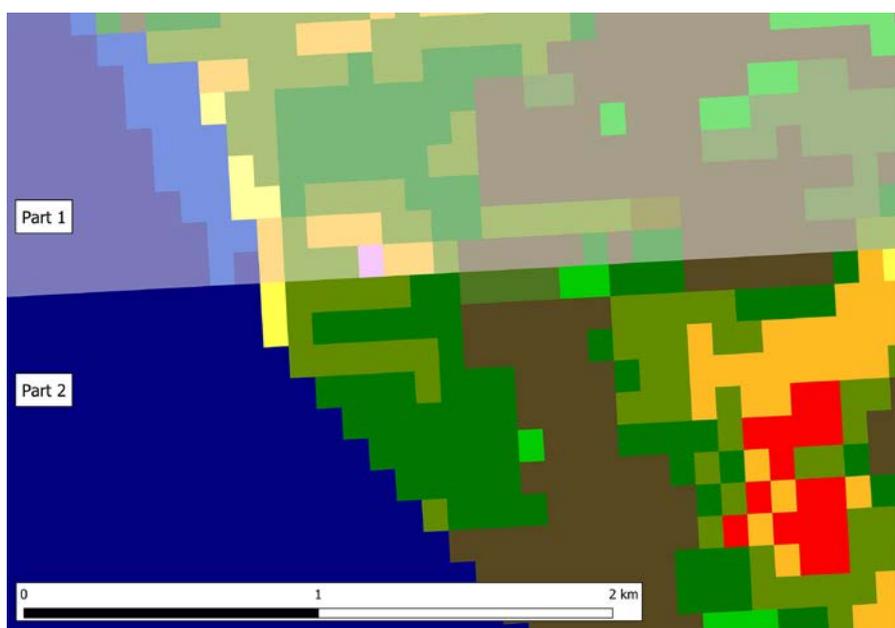


Fig. 7 CGLS-LC100 map's section alignment.

problem (Fig. 7), despite the fact that this map was also downloaded in the same way, by individual sections. On such a small scale (30 m), these differences are significant and will affect the results of the analysis.

FROM-GLC10 map (Gong et al. 2019)

This map is the first freely available global land cover map with a resolution of 10 m. Map for 2015 has regional level classification, map for 2017 only global classification (Table 7). This map's sections are well aligned, with only one pixel separating them. The colouring of the FROM-GLC10 map is the same as that of the ESA CCI-LC map and some regional level groups have the same colour as the global level class (Fig. 8a), making it difficult to estimate the level of detail of this map online. In this particular area this map tends to classify sparse herbaceous vegetation or shrubland as "Natural grassland". As there is no explanation of classification parameters or comparison with the LCCS system, it is difficult to determine whether this map classifies this type of vegetation correctly.

Conclusions

At a global scale, study maps should be compared for several parts of the main target area of a study. For correct comparison, these maps should be downloaded and then carefully investigated. Some maps may provide a better representation of particular classes than others. Note that the ideal classification of a large number of classes at a small resolution is almost unachievable, but such minor misclassifications can be easily fixed.

During this investigation it was observed that for the particular area studied, most maps appear to misclassify sparse mosaic shrub and herbaceous vegetation (prob-

Table 7 Regional and global classification for the FROM-GLC10 map, adapted from Gong et al. (2019).

Map land cover class	
Global level classification	Regional level classification
Cropland	Rice paddy
	Greenhouse
	Orchard
	Bare farmland
	Other
Forest	Broadleaved-on
	Broadleaved-off
	Needles-on
	Needles-off
	Mixed leaved-on
	Mixed leaved-off
Grassland	Natural grassland
	Desiccated
	Pasture
Shrubland	Shrubland, leaves-on
	Shrubland, leaves-off
Wetland	Marshland
	Marshland, leaves-off
	Mudflat
Tundra	Shrub and brush tundra
	Herbaceous tundra
Snow/Ice	Snow
	Ice
Water	
Impervious surface	
Bare land	
Cloud	

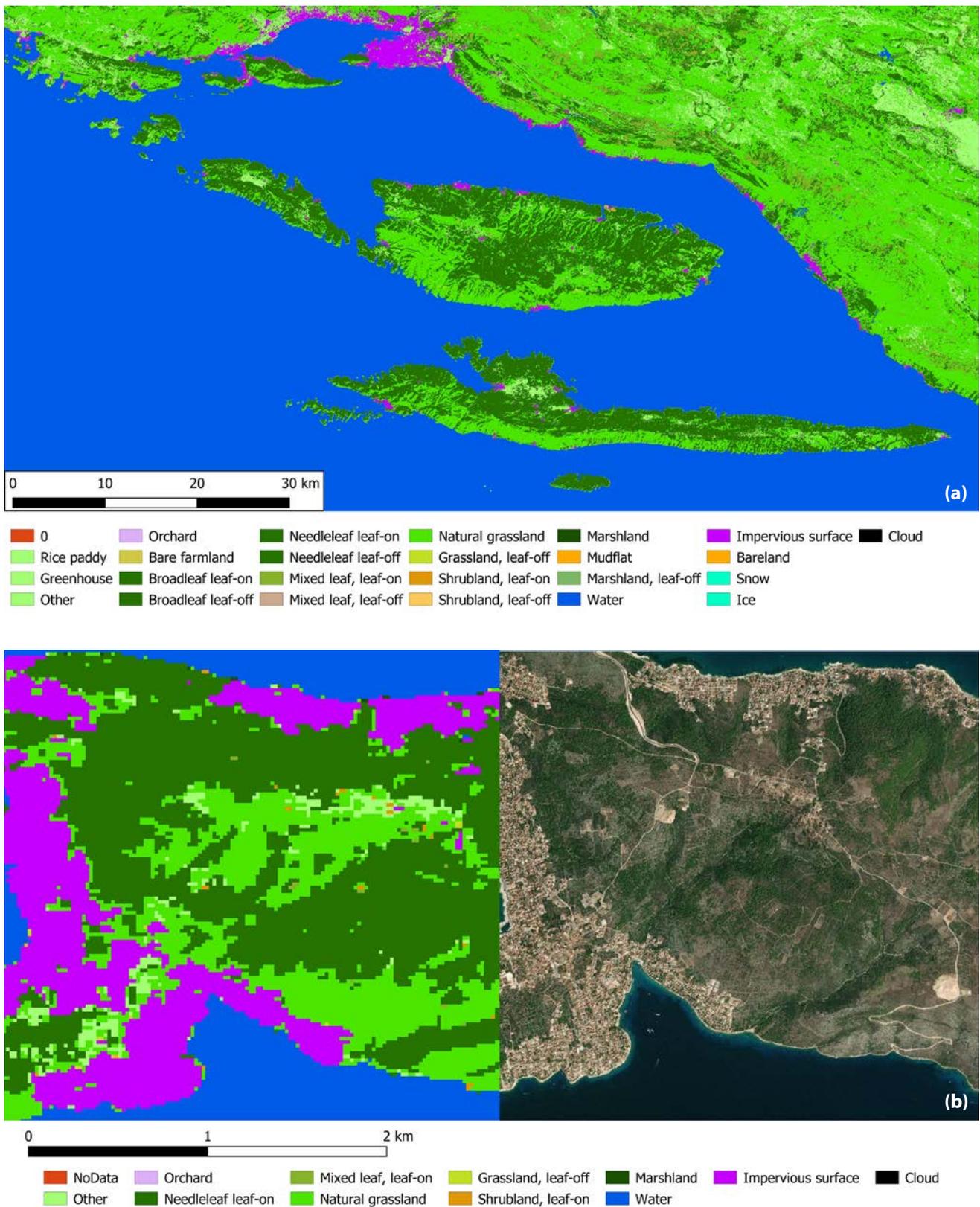


Fig. 8 FROM-GLC10 map, (a) less detailed, legend includes all map classes; (b) more detailed, only those classes present in area are shown.

ably perennial) mixed with bare soil (rocks), visible on satellite imagery. The Copernicus CGLS-LC map tends to indicate that this area is covered by crops or cultivated and managed vegetation, despite the fact that it has more suitable classes. The ESA CCI-LC map, on the other

hand, classifies these areas as “Cropland rainfed”, which is mostly correct; or incorrect as “Tree cover broadleaved deciduous closed to open (> 15%)”. However, this could be a problem unique to this area that requires further investigation. The GLOBCOVER map provides a mislead-

ing classification of particular types of vegetation (forest) and incorrect classification of urban areas, which make this map unsuitable for use for studying this area. The resolution of map GLC-SHARE from FAO is too coarse for evaluating habitat suitability as it does not describe the pattern of vegetation or even the shape of the mainland. There may be another version of the GLC_FCS30 map (Liangyun et al. 2020) that is properly aligned and has averaged classification for overlapping parts, which is more accurate. FROM-GLC10 map (Gong et al. 2019) is the first global map that is freely available with such small resolution. This map has some noticeable artefacts due to its resolution, 10 m, however, such minor artefacts can be removed. ESA CCI-LC and Copernicus CGLS-LC100 maps provide the most accurate estimates for the area studied. Incorrect position of classes CCI-LC map against satellite imagery (Gorelick et al. 2017) may be due to the level of resolution. Both maps sometimes misclassify certain types of vegetation (sparse vegetation on bare soil), but if this is consistent and exclusive to this region, it can be manually corrected.

This comparison indicates that the Copernicus CGLS-LC100 and ESA CCI-LC maps seem to be the most universal maps for determining potentially suitable habitats. They have a wide range of land cover groups that can be compared (using LCCS classification system), making it easier to decide which spatial resolution to use.

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