



Review

Urban phenological studies – Past, present, future

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ABSTRACT

Phenology is believed to be a suitable bio-indicator to track climate change. Based on the strong statistical association between phenology and temperature phenological observations provide an inexpensive means for the temporal and spatial analysis of the urban heat island. However, other environmental factors might also weaken this relationship. In addition, the investigation of urban phenology allows an estimation of future phenology from current information since cities with their amplified temperatures may serve as a proxy for future conditions. Nevertheless, the design of spatial compared to long-term studies might be influenced by different factors which should be taken into consideration when interpreting results from a specific study. In general, plants located in urban areas tend to flush and bloom earlier than in the countryside. What are the consequences of these urban–rural differences? This review will document existing findings on urban phenology and will highlight areas in which further research is needed.

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1. Introduction

One of the most central problems the world is currently facing is anthropogenic climate change. During the period 1880–2012 mean global air temperature increased by 0.85 [0.65 to 1.06] °C which is very likely attributable to the increase in anthropogenic greenhouse gases, with carbon dioxide (CO₂) being the most important driver (IPCC, 2013). Although global temperature change has been rather flat for the last decade (Hansen et al., 2013), temperature will – depending on the underlying scenario – increase between 1.0 and 3.7 [0.3 to 4.8] °C by the end of the 21st century (IPCC, 2013).

Implications of global climate change, particularly related to the increase in air temperature, can already be seen in various biological systems (Parmesan and Yohe, 2003; Root et al., 2003; Rosenzweig et al., 2008; IPCC, 2013). One of the best bio-indicators of climate change is phenology (Walther et al., 2002; Badeck et al., 2004; Cleland et al., 2007), the timing of natural recurring events such as leaf unfolding, flowering, fruit ripening or leaf colouring and fall. The strong relationship between spring phenology and air temperature in temperate regions allows

observed changes in phenology to be related to changes in temperature (Sparks and Carey, 1995; Menzel, 1997; Menzel and Fabian, 1999; Menzel et al., 2006).

Human activities do not only modify climatic conditions at the global scale by altering the composition of atmospheric gases; they are also involved in a much more localised phenomenon – the urban heat island (UHI) effect that is associated with higher temperatures in cities (Landsberg, 1981; Fezer, 1995; Matzarakis, 2001). This temperature increase is caused by the replacement of natural landscapes with impermeable surfaces and built up areas. Unlike natural environments, urban areas are mostly characterised by highly compacted or impermeable surfaces with distinctive differences in heat capacity and conductivity (Landsberg, 1981). The relatively low abundance of vegetation implies less evapotranspiration and therefore higher temperatures (Oke, 1976; Landsberg, 1981). In addition to the common perception that the reduction in evaporative cooling is mainly responsible for the UHI effect (Taha, 1997), it was recently found that the efficiency of heat convection to the lower atmosphere, which depends on the local background climate, is also of major importance (Zhao et al., 2014). The emission of sensible heat along with latent heat and chemical generation of moisture associated with private and industrial energy consumption are additional factors contributing to increased urban temperatures (Sailor, 2011). Urban air pollutants have an effect on energy fluxes: particulate matter absorbs/scatters the incoming solar radiation resulting in a decrease of global radiation

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and an increase in diffuse radiation (Landsberg, 1981; Kuttler, 2004). Absorption and re-emission by infrared active gases and aerosols within urban air lead to higher downward longwave atmospheric radiation (Kuttler, 2004).

The magnitude of the temperature difference between the city and its countryside is influenced by prevailing synoptic weather conditions (Landsberg, 1981): clear and calm nights during winter are associated with the highest excess heat. Moreover, the UHI depends on the size of the city and its building density (Landsberg, 1981; Zhang et al., 2004). The UHI effect has been well documented for numerous cities in different parts of the world (Landsberg, 1981; Bründl et al., 1987; Matzarakis, 2001) ranging on average from +0.5 to +3.0 °C with maximum values of +15 °C during specific weather conditions (Kuttler, 2004).

2. Scope of urban phenology studies

Studies on urban phenology are useful in several ways. First, they allow for the detection of urban heat islands. Second, they can be used for the assessment of climate change impacts on phenology (Fig. 1). Furthermore, urban phenology is valuable in citizen science projects since people who participate in phenological observations will learn more about ecological processes and changes while simultaneously awareness on climate change and its impacts will be strengthened (Butler and MacGregor, 2003; Gazal et al., 2008).

2.1. Detection of urban heat islands

Based on the statistical association between phenology and temperature (Cannell and Smith, 1986; Wielgolaski, 1999; Menzel and Fabian, 1999), phenological observations provide a suitable and inexpensive basis for the temporal and spatial analysis of the urban heat island (e.g., Omoto and Aono, 1990; Bernhofer, 1991; Lakatos and Gulyás, 2003; Matsumoto et al., 2009).

Phenological observations for assessing microclimatic conditions have a long tradition in temperate cities. Numerous broad-scale (i.e., spatial/local) and long-term studies exist on phenology in urban areas, especially in Europe, North America and Asia (see Table 1). Broad-scale studies are characterised by a dense coverage of a restricted region and are often based on a single or a few observation years (e.g., Mimet et al., 2009; Fukuoka and Matsumoto, 2003; Lakatos and Gulyás, 2003; Jochner et al., 2013a). Phenological observations involving citizen science (with school children, students or other volunteers) can result in a high number of observation sites and may imply a greater geographical extent (e.g., Henniges and Chmielewski, 2006; Gazal et al., 2008). Long-term studies (e.g., Rötzer and Sachweh, 1995; Defila, 1999; Rötzer et al., 2000; Jochner et al., 2012) which are mostly based on data derived from phenological networks focus on longer periods but mostly incorporate fewer observation sites where preferably always the same individual is observed. Phenological studies do not just include ground observations at the species level but also remote sensing referring to the ecosystem level (e.g., White et al., 2002; Fisher et al., 2006). In addition, herbarium data which date back several decades or centuries can be utilised to detect

phenological changes in urban and rural environments (e.g., Primack et al., 2004; Lavoie and Lachance, 2006; Neil et al., 2010).

2.1.1. Urban–rural differences

A simple method to describe the effect of the UHI on phenology is the investigation of differences between the city and the countryside (Rötzer and Sachweh, 1995). In general, plants tend to develop earlier in the cities with an advance from a few days up to a couple of weeks compared to their rural surroundings (see Table 1). Zhang et al. (2004) reported that the UHI effect on plant phenology was stronger in North America than in Europe or Asia; a fact probably related to the dense and vertical urban design of North American cities causing higher urban temperatures (Zhang et al., 2004; Bonan, 2002). In addition, different species or different phenophases within the same species respond differently to urban climate. Herbaceous plants are more sensitive to microclimatic variation than trees (Mimet et al., 2009); early phases are known to be more sensitive to temperature (Fitter and Fitter, 2002) and are additionally influenced by the amplified UHI effect in winter (Rötzer and Sachweh, 1995; Defila, 1999; Rötzer et al., 2000; Jochner et al., 2012). A recent study demonstrated that first flowering dates of wind-pollinated species were associated with greater advances than insect-pollinated species (Ziello et al., 2012a). This is in contrast to the finding of Fitter and Fitter (2002) who reported greater advances for insect-pollinated plants. Both these studies, however, were not specifically based on urban–rural comparisons. Detailed investigations related to the differences in pollination types in urban areas are still lacking and further research is required to reveal likely different responses.

Unlike spring and summer phenophases, autumn phenophases are not always associated with distinct urban–rural differences: Defila (1999) reported no differences for autumn phenophases in urban and rural sites of Zurich, Switzerland. Rötzer (2007) documented a four day advance of leaf colouring of horse chestnut (*Aesculus hippocastanum*) in cities in Bavaria, Germany, which, however, might be influenced by the leafmining moth. Conversely, International Phenological Garden (IPG) data revealed that densely populated areas were linked to a delay of five days for leaf colouring and leaf fall (Rötzer, 2007). In urban areas of Japan, leaf colouring of gingko (*Ginkgo biloba*) and Japanese maple (*Acer palmatum*) occurred up to three weeks and ten days later, respectively (Fukuoka and Matsumoto, 2003). Rötzer (2007) suggested that less densely built areas were linked to a delay in autumn onset dates due to beneficial high temperatures; plants in extreme densely built areas, however, were susceptible to stressful environmental conditions such as increased pollution or soil water deficit and might respond by earlier leaf fall. This is also in agreement with Ziska et al. (2003) who documented earlier senescence for common ragweed (*Ambrosia artemisiifolia*) in urban Baltimore, USA. These results show that there is still a lack of knowledge how autumn phenophases in cities are influenced by environmental factors and further urban studies should include autumn phenological events.

Numerous studies using satellite/space borne remote sensing reported a prolongation of the growing season in urban areas, mostly attributable to an earlier start of the season. For example

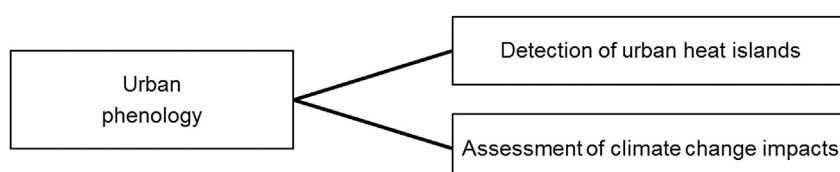


Fig. 1. Application fields of urban phenology.

Table 1

Overview of urban phenology studies. F = flowering, BB = budburst, LU = leaf unfolding, LC = leaf colouring, LF = leaf fall, PL = planting, H = harvest; negative values in the column "Main results" (unless other specified) refer to; negative/positive values to an advance/delay in the city.

Author(s)	Year	Survey area	Observed plants	Period	Main results
Baumgartner	1952	Munich, Germany	16 spring/summer phenophases	1952	Max. intra-urban differences of −9 to −37 days
Franken	1955	Hamburg, Germany	<i>Forsythia suspensa</i> (F)	1955	Up to −17 days
Zacharias	1972	Berlin, Germany	<i>Tilia × euchlora</i> (F)	1967	Up to −10 days
Baumgartner et al.	1984	Munich, Germany	<i>Aesculus hippocastanum</i> (LU, F), <i>Tilia platyphyllos</i> (LU, F), <i>Forsythia viridissima</i> (LU, F), <i>Cornus mas</i> (LU, F)	1984	Considerable small-scale differences, representative for thermal differences
Karsten	1986	Mannheim, Germany	<i>Robinia pseudoacacia</i> (LU, F), <i>Forsythia suspensa</i> (F), <i>Platanus acerifolia</i> (LU, F)	1978 −1983	Distinctive small-scale differences, phenology matches the microclimatic structure
Koch	1986	Vienna, Austria	<i>Prunus avium</i> (F), <i>Malus domestica</i> (F), <i>Prunus armeniaca</i> (F), <i>Pyrus communis</i> (F)	1973 −1982	−10 days to −13 days
Omoto and Aono	1990	several Japanese cities	<i>Prunus yedoensis</i> (F)	1965 −1983	−0.2 days to −7.8 days
Bernhofer	1991	Vienna, Austria	<i>Forsythia suspensa</i> (F, LU)	1998	−16 days to −29 days
Rötzer and Sachweh	1995	Munich, Germany	<i>Galanthus nivalis</i> (F), <i>Forsythia</i> sp. (F), <i>Prunus avium</i> (F), <i>Malus domestica</i> (F)	1951 −1990	−5 to −18 days
Rötzer	1996	Munich, Nuremberg, Augsburg and Regensburg, Germany	8 phenological phases	1951 −1980	Spring: −3.5 days to −7.5 days, autumn: +3.8 days
Aono	1997	Osaka, Kyoto and Kobe, Japan	<i>Prunus yedoensis</i> (F)	−1950 −1990	−1.4 days (1950s), −5.4 days (1980s) (Kobe) −3 days to −5 days (Osaka 1989) −3.5 days to −8.0 days
Rötzer et al.	1997	Munich, Nuremberg, Augsburg and Regensburg, Germany	8 phenological phases	1961 −1990	
Sachweh and Rötzer	1997	Munich, Nuremberg, Augsburg and Regensburg, Germany	<i>Galanthus nivalis</i> (F), <i>Forsythia</i> sp. (F), <i>Prunus avium</i> (F), <i>Malus domestica</i> (F)	1951 −1990	Remarkably strong trend for <i>Galanthus nivalis</i> in urban areas (−7.3 days/decade)
Defila	1999	Zurich, Switzerland	28 phenophases	1986 −1995	−1 week to −2 weeks
Rötzer et al.	2000	10 central European cities	<i>Galanthus nivalis</i> (F), <i>Forsythia</i> sp. (F), <i>Prunus avium</i> (F), <i>Malus domestica</i> (F)	1951 −1995	−1.5 days to −4 days, stronger trends (1980–1995) in rural (up to −15.3 days/decade) compared to urban areas (up to −13.9 days/decade)
Baker et al.	2002	Phoenix, USA	<i>Gossypium</i> (PL, H)	1997 −2000	−14 days (PL), −22 days (H)
White et al.	2002	Eastern USA	ecosystem level (AVHRR NDVI)	1990 −1999	Vegetation period +7.6 days; start of vegetation −5.7 days
Defila & Clot	2003	Geneva and Liestal, Zurich	<i>Aesculus hippocastanum</i> (BB), <i>Prunus avium</i> (F)	1808 −2002 1894 −2002	Stronger urban trend (−2.4 days/decade) compared to rural trend (−0.8 days/decade)
Fukuoka & Matsumoto	2003	Kumagaya City, Japan	<i>Prunus yedoensis</i> (F), <i>Ginkgo biloba</i> (LC), <i>Acer palmatum</i> (LC)	2000 −2001	−2 days (<i>Prunus</i>), +2 weeks to 3 weeks (<i>Ginkgo</i>), +7 days to 10 days (<i>Acer</i>)
Lakatos & Gulyás	2003	Debreczin and Szegedin, Hungary	<i>Forsythia suspensa</i> (F)	2003	−4 days to −8 days
Ziska et al.	2003	Baltimore, USA	<i>Ambrosia artemisiifolia</i> (F)	2000 −2001	−3 days to −4 days
Primack et al.	2004	Boston, USA	372 records of herbarium specimen (37 genera)	1885 −2003	More visible trend towards earlier flowering due to urban heat island effect
Zhang et al.	2004	North America, Europe and Asia	ecosystem level (MODIS data)	2001	Start of vegetation 4 days to 9 days earlier, end 2 days to 16 days later (North America)
Fisher et al.	2006	Providence, Eastern USA	ecosystem level (Landsat TM, ETM+)	1984 −2002	−5 days to −7 days earlier start of vegetation
Henniges and Chmielewski	2006	Berlin, Germany	11 spring phenophases	2006	−0.1 days to −7.1 days
Lavoie and Lachance	2006	Montreal and Quebec City, Canada	<i>Tussilago farfara</i> (F)	1918 −2003	Earlier flowering only noticeable in large urban areas
Lu et al.	2006	Beijing, China	<i>Prunus davidiana</i> (F), <i>Prunus armeniaca</i> (F), <i>Robinia pseudoacacia</i> (F), <i>Syringa oblata</i> (F)	1950 −2004	−1.5 days/decade to −2.9 days/decade
Luo et al.	2007	Beijing, China	<i>Prunus davidiana</i> , <i>Hibiscus syriacus</i> , <i>Cercis chinensis</i> (all BB, F, LC, LF)	1962 −2004	Pronounced changes with time in response to climate change and UHI
Rötzer	2007	IPG, Europe	different tree and shrub species	1961 −1998	−2.1 days (LU), −4.8 days (F), +4.8 days (LC), +5.4 days (LF)
Gazal et al.	2008	7 cities in Asia, Europe, Africa and North America	7 species (BB)	2005	−23 days to +9 days
Matsumoto et al.	2009	Tokyo, Japan	<i>Prunus yedoensis</i> (F)	2004	−5 days to −6 days
Mimet et al.	2009	Rennes, France		2005	

Table 1 (continued)

Author(s)	Year	Survey area	Observed plants	Period	Main results
			<i>Platanus acerifolia</i> (BB, F), <i>Prunus cerasus</i> (BB, F)		Urban–rural gradient in onset dates, influential factors: DTR, Tmin, ground cover types
Shustack et al.	2009	Central Ohio, USA	<i>Aesculus glabra</i> (LU), <i>Lonicera maackii</i> (LU), <i>Acer negundo</i> (LU)	2006	Abundance of exotic species caused phenological urban–rural differences
Neil et al.	2010	Phoenix metropolitan area, Arizona, USA	87 shrubs and ephemerals (F)	1902–2006	28% of species with significant urban–rural differences, 95% associated with earlier urban onsets
Fotiou et al.	2011	Thessaloniki, Greece	<i>Parietaria judaica</i> (F)	2005	Intra-city differences of –12 days (spring) and +3 days (autumn) between a south and a north-facing population
Jeong et al.	2011	9 cities in South Korea	<i>Forsythia koreana</i> , <i>Rhododendron mucronulatum</i> , <i>Prunus yedoensis</i> , <i>Prunus persica</i> (all F)	1954–2004	Degree of advancement is roughly proportional to degree of urbanisation
Jochner et al.	2011	Munich and Ingolstadt, Germany	<i>Betula pendula</i> (F, LU), <i>Aesculus hippocastanum</i> (F), <i>Corylus avellana</i> (F), <i>Forsythia suspensa</i> (F)	1951–2008; 2009	High temperatures in spring lead to less pronounced urban–rural differences in onset dates and vice versa
Jochner et al.	2012	Frankfurt, Cologne and Munich, Germany	9 spring phenophases	1980–2009	Changes in land use from rural to urban leads to advances of 2.6 days to 7.6 days; temporal trends were not affected by urbanisation
Beaubien	2013	Edmonton, Canada	7 spring phenophases (F)	1931–1961, 1987–2006	Significant earlier urban onset for 5 phenophases (–1.7 days to 4.5 days)
Han and Xu	2013	Six cities in the Yangtze River Delta, China	ecosystem level (SPOT/VGT NDVI)	2002–2009	Vegetation period +12 days; start of vegetation –5.9 days
Jochner et al.	2013c	Munich, Germany	<i>Betula pendula</i> (F, LU)	2010–2011	Air temperature explained between 55.7 and 83.7 % of the variance in onset dates; influence of nutrients
Jochner et al.	2013a	Munich, Germany and Campinas, Brazil	<i>Betula pendula</i> , <i>Tabebuia chrysotricha</i> , <i>Caesalpinia pluviosa</i> , <i>Tipuana tipu</i> (all F, LU)	2011	Only <i>Tipuana tipu</i> was temperature sensitive and flowered significantly earlier in the city (–19.7 days)
Jochner et al.	2013b	Munich and Ingolstadt, several sites in Bavaria, Germany	<i>Betula pendula</i> (F)	2010–2011	Temperature response rates were significantly higher for spatial (–4.4 days/°C) compared to long-term data (–1.9 days/°C)
Tryjanowski et al.	2013	Western Poland	18 migrant birds	1983–2010	Mean: +3 days; stronger mean urban (–0.62 days/year) than rural trend (–0.31 days/year)
Comber and Brunsdon	2014	England and Wales	<i>Crataegus monogyna</i> , <i>Prunus spinosa</i> , <i>Aesculus hippocastanum</i> (F)	1934–2007	10% increase in urban land use advances phenophases by 0.56 days to 1.19 days

(see also Table 1), White et al. (2002) documented a prolongation of the vegetation period in urban deciduous broadleaf forests in the eastern United States by eight days. Even a 15 day longer growing season in cities across the eastern United States was documented by Zhang et al. (2004). Han and Xu (2013) reported an expanded vegetation period by an average 12 days for six cities in the Yangtze River Delta, China. The number of studies using satellite derived data in Europe is, however, rare and further research should also concentrate on Europe where an abundance of *in situ* observations can be used to compare the results derived from these two different techniques.

Apart from plant phenological studies other abiotic and biotic phenological studies are rather exceptional. Recently, Townroe and Callaghan (2014) demonstrated that mosquitos breeding in urban experimental water butts in South East England were associated with earlier occurrence and peak dates in larval abundance compared to rural containers. It is known that higher water temperature advances larval development (Loetti et al., 2011) and the phenology of insects in general might be positively affected by climate change and thus higher urban temperatures (Bale et al., 2002). Miller-Rushing et al. (2008) reported that urbanisation probably led to earlier arrival dates of early migration birds in Cambridge, Massachusetts (USA). However, Tryjanowski et al.

(2013) analysed avian phenology in urban and rural areas in western Poland and reported a generally earlier arrival of migratory birds in rural areas. This somewhat unexpected finding is indicative of the need for further studies investigating phenological behaviour of birds and other animals.

2.1.2. Influential and confounding factors

2.1.2.1. Temperature: from means to extremes. Urban–rural differences in phenology can vary across years and cannot entirely be explained by differences in temperature between the city and the countryside (Jochner et al., 2011). In general, temperature is able to explain more than two thirds of the variance in phenological onset dates (Menzel and Fabian, 1999). However, broad-scale urban studies often reported relatively modest R^2 values: Matsumoto et al. (2009) calculated an R^2 value of 51.8% derived from a linear model of onset dates of cherry (*Prunus yedoensis*) flowering in Tokyo, Japan, on mean March temperature in 2004. Lakatos and Gulyás (2003) reported an R^2 value of 41.2% for onset dates of full flowering of forsythia (*Forsythia suspensa*) on UHI intensity in Debrecen, Hungary, in spring 2003. Jochner et al. (2013b) found that mean air temperature in March and April explained between 55.7 and 83.7% of the variance in onset dates of birch (*Betula pendula*) phenophases in 2010 and 2011 in Munich and Ingolstadt,

Germany. This indicates that, in addition to mean air temperature of the preceding months, other temperature variables or influential factors might be important when analysing urban phenology.

The diurnal temperature range (DTR) which is smaller in urban compared to rural areas is a good measure for the UHI effect. Mimet et al. (2009) found that DTR was best correlated with budburst onset dates of sour cherry (*Prunus cerasus*) in the metropolitan area of Rennes, France. Since the UHI effect is most pronounced at night (Landsberg, 1981; Baker et al., 2002; Mimet et al., 2009; Shustack et al., 2009) minimum or night time temperatures, which are highly correlated with onset dates, are able to adequately describe urban excess heat (Jochner et al., 2013a). In contrast, maximum temperature was not found to be of great importance in phenological studies (Wielgolaski, 1999; Mimet et al., 2009; Jochner et al., 2013a).

However, the prevailing temperature conditions shortly before flowering can mask the UHI effect on phenology. Baumgartner (1952) proposed that intra-urban differences in phenology were more pronounced when spring temperatures rise gradually. A study by Jochner et al. (2011) showed that warm or cold spells can indeed influence phenological behaviour at the local scale: High temperatures shortly before the phenological onset of a specific event are likely to result in a simultaneous onset; low temperatures rather contribute to a larger delay between urban and rural onset dates. In general, extreme weather conditions might have greater effects on phenology than changes in mean temperatures (Jentsch et al., 2007). The frequency and intensity of warm spells are very likely to increase in the future (Schär et al., 2004; IPCC, 2013). Therefore, it can be assumed that warm spells in spring might lead to smaller urban–rural differences in phenology. However, there is still a need to further investigate the phenological behaviour under extreme temperatures to assess the validity of this assumption.

As a result of its thermal inertia soil temperature (which reflects accumulated chill and heat units) might also be a useful predictor in phenological research (Baldocchi et al., 2005). The usefulness of soil temperature was already mentioned by Taylor (1974) who observed that leaf-out in Tennessee occurred when the soil temperature had reached the level of the mean annual air temperature of this region. In addition, Baldocchi et al. (2005) found that the onset of net carbon uptake by temperate deciduous forests corresponds with the date when soil temperature equals annual mean air temperature. However, the authors also emphasised that the suitability of soil temperature may not apply for all functional plant types and climatic zones such as boreal biomes as reported by Suni et al. (2003). Since there are no studies analysing the impact of urban soil temperature or urban–rural differences of soil temperature on plant phenology further research should specifically examine whether this meteorological variable is useful in predicting onset dates also in urban environments.

2.1.2.2. Photoperiod and chilling. In addition to the prominent role of temperature, there are also modifications by photoperiod (Leopold, 1951; Levy and Dean, 1998; Schaber and Badeck, 2003; Caffarra et al., 2011a,b). Its importance is species-specific and has been demonstrated in experimental studies (Saxe et al., 2001; Caffarra et al., 2011b) and phenological models (Schaber and Badeck, 2003; Caffarra et al., 2011a). The peculiarity of urban areas in this context is the pollution by light, e.g., by streetlights, buildings (exterior and interior lighting) or illuminated advertising which might exert an influence on plants growing in the city. However, to date there exists no study analysing the effect of light pollution on plant phenology (Neil and Wu, 2006).

Furthermore, plant-specific chilling requirements are an important factor for phenology (Chuine et al., 2010). In general, plants have to experience cold conditions in autumn/winter to

break dormancy. Sufficient chilling is expected to decrease the amount of forcing temperatures needed in spring. In turn, warm autumn and winter months with insufficient chilling may result in later phenological onset dates (Murray et al., 1989; Partanen et al., 1998; Heide, 2003). Using an extensive experimental setup, Laube et al. (2014) found that the effect of chilling greatly exceeded that of photoperiod; however, pioneer or invasive species were found to have lower chilling requirements. As the UHI is most pronounced in winter months, urban areas are especially affected by high winter temperatures (Gazal et al., 2008), and the inclusion of chilling sums is highly recommended within urban phenological studies. However, detailed investigations about differences in the chilling fulfilments in urban and corresponding rural areas are lacking and further research should focus on this topic.

2.1.2.3. Nutrients. Chemical and physical soil characteristics are also known to influence phenology (Wielgolaski, 2001). However, phenological studies often only analyse the effects of nutrient supply by manipulation experiments adding fertilisers to the soil or by the actual assessment of the availability of nutrients in the soil (Wielgolaski, 2001). Existing studies either found no effect of nutrients (Roberntz, 1999; Amundson et al., 1995) or significant changes (either delays or advances) in phenology (Chandler and Dale, 1990; Amundson et al., 1995; Wielgolaski, 2001). Conventional soil analyses, however, do not account for the heterogeneous conditions of urban soil and impervious surface might completely obstruct such analyses. Therefore, Jochner et al. (2013c) analysed foliar nutrient concentration of birch leaves in Munich in 2010 and 2011. Increases in the concentration of potassium, zinc, boron and calcium, which influence cell extension as well as membrane function and stability (Marschner, 1995), showed a positive and statistically significant association with phenological onset dates. Potassium had the strongest influence on phenology, especially on leaf unfolding, and was even found to be a significant predictor (in addition to temperature as the main trigger) within linear mixed models. Therefore, using potassium was sufficient to explain phenological variations. This, however, might not be the case in different regions with different soil conditions and nutrient availability. Higher temperatures – via altered soil moisture – lead to higher mineralisation rates and therefore affect translocation rates and nutrient availability (White and Haydock, 1970; Fisher, 1980; Nord and Lynch, 2009; Marschner, 1995). We suggest that foliar nutrient concentrations reflect the physiological performance of a tree more adequately than soil concentrations. However, an explicit comparison between soil and foliar nutrients in urban trees does not exist. This question should be explored in further research. Moreover, it is recommended that an analysis should be extended to a wider range of plant species, soil conditions, and regions to obtain more general information about the influential character of single nutrients.

2.1.2.4. Other factors. There exists an array of external and internal factors influencing phenology such as precipitation, air humidity, diseases, pests, competition, pollutants, individual genes and age (Menzel, 1999; Menzel et al., 2008). It has also been suggested that hydrological modification in urban areas such as increases in runoff and decreases in groundwater might affect phenology (Han and Xu, 2013). However, detailed investigations are lacking and are needed to fully understand the effect of urban environmental conditions on plant phenology.

2.1.3. Regional peculiarities

Urban phenology studies in tropical cities are not only rare; phenological behaviour of tropical trees is also not adequately understood yet (Borchert et al., 2002; Singh and Kushwaha, 2005). Tropical phenology is regarded as a less adequate indicator of global

warming (Borchert et al., 2005), since phenology is also influenced by other parameters such as leaf longevity, precipitation, ground-water availability, seasonal variation in water stress and day length (Borchert, 1994; Eamus and Prior, 2001; Borchert et al., 2005; Do et al., 2005; Singh and Kushwaha, 2005). Gazal et al. (2008) reported that budburst in the tropical city of Bangkok (Thailand) was advanced (−23 days) but delayed (+9 days) in Korat (Thailand) and Dakar (Senegal). Furthermore, they documented a high variation in onset dates of phenological phases in the tropics. This was also found by Jochner et al. (2013a) analysing tropical tree phenology in Campinas, Brazil. The authors found mostly non-significant urban–rural differences which might be partly attributable to the high within-species variation in tropical trees. Significant earlier urban onset dates were only demonstrated for racehorse tree (*Tipuana tipu*) and ranged between −10.5 and −19.7 days. Nearly all analysed phenophases of the other two tropical trees, yellow trumpet tree (*Tabebuia chrysotricha*) and false Brazil wood (*Caesalpinia peltopheroides*), even showed delayed urban onset dates (although not significant). Leaf unfolding phenophases of racehorse tree showed high correlations with air temperature and are therefore suitable for urban phenology studies. However, the huge number of potential plants that could be monitored in tropical cities makes it necessary to further explore additional species and phenophases. In future research, other parameters which are also likely to influence phenology (precipitation, air humidity, edaphic factors, leaf longevity, day length) should also be implemented since they might be helpful predictors in tropical phenology.

In addition to tropical regions, urban phenological studies in arid environments have also been neglected (Neil and Wu, 2006). In recent years, however, some urban studies in arid regions have been published: using NDVI (Normalized Difference Vegetation Index) data, Buyantuyev and Wu (2012) analysed the effect of urbanisation on phenology and its association to climate variables in the Phoenix metropolitan region, USA, a hot and dry (<200 mm) region with two yearly precipitation maxima. In response to urbanisation they found a complex and diverse pattern of land surface phenology which was uncorrelated to climate variability. The authors argued that the dominance of evergreen and large woody species as well as active maintenance in cities was responsible for a comparably long growing season. Urban communities which are composed of plants of different evolutionary histories and physiological mechanisms make it difficult to predict phenological patterns at the ecosystem and landscape level. Therefore, Buyantuyev and Wu (2012) indicated that not only environmental cues such as photoperiod and climate affect urban–rural differences, but also ontogenetic mechanisms at the genetic and physiological level as well as socio-economic drivers related to land management (e.g., landscaping and water irrigation) have to be considered. The influence of exotic species on satellite derived phenological metrics was also mentioned by Shustack et al. (2009) who analysed leaf phenology of trees in 11 forests spanning an urban–rural gradient in Central Ohio, USA. They found that the abundance of exotic species implied an earlier start of the vegetation period and warned that differences in greenup across large areas should be interpreted with caution.

The influence of urban climate on lowland desert plants in the Phoenix metropolitan area was investigated by Neil et al. (2010) using herbarium records. They found that only 28% of the selected species showed differences in flowering times between urban and non-urban areas with even 5% showing delayed onset dates in cities. Since the authors did not analyse climate data, causative conclusions on these findings cannot be drawn. Focussing on arid environments, Jenerette et al. (2013) analysed the growing season length of seven metropolitan regions in the southwestern United States using satellite derived data. They found a lengthening of the growing season in urbanised areas in five cases and

attributed differences to irrigation management and planting of non-native species. They emphasised the importance of precipitation for arid environments and suggested that future studies should look towards understanding whole metropolitan water fluxes.

2.2. Assessment of climate change impacts on plants

Since the magnitude of the UHI in many cities is comparable to future projected temperature increase at the global scale (1.0–3.7 °C; IPCC, 2013) urban areas can be regarded as a proxy exhibiting conditions that are expected for more rural environments in the future (Ziska et al., 2003; Zhang et al., 2004; Luo et al., 2007; Mimet et al., 2009). Thus, phenological differences between the city and the countryside can be used for more than temporal and spatial analysis of the urban heat island effect; these differences also provide evidence on how phenology will respond to future temperature increase and to alterations of other environmental conditions typical for the urban climate. The majority of studies on climate change impacts on phenology focused on non-urban ecosystems (Neil and Wu, 2006). In general, it is necessary to distinguish between the effects of local urban warming and the more general characteristics of global climate change (Chung et al., 2009; Primack et al., 2009; Beaubien, 2013). Understanding the impact of the anthropogenic UHI effect on phenology allows a better assessment of the impact of climate change *per se* and counteracts incorrect interpretations of phenological data and trends (Defila, 1999). The approach of using urban–rural gradients as an empirical analogue (Beggs, 2004) is an appropriate and low-cost alternative to open-top chambers, climate chambers and free-air CO₂ enrichments (Ziska et al., 2003).

2.2.1. Temporal urban and rural trends

The analysis of temporal trends (changes in days per time unit) is a simple way to assess the impacts of climate change on phenology. The extrapolation of these trends enables the estimation of future phenological changes. The comparison of urban and rural trends in particular facilitates an evaluation of phenological changes under different temperature conditions.

Some authors (e.g., Lu et al., 2006; Luo et al., 2007) only analysed phenological time series of urban areas and did not compare these results with rural stations. Therefore, temporal trends reported in these studies are the consequence of both large-scale climate warming and the effect of the UHI. Jeong et al. (2011), for example, did not include data of the respective countryside, but incorporated nine differently urbanised cities in South Korea to assess changes depending on the degree of urbanisation.

Rötzer et al. (2000) reported that phenological trends (1980–1990) of spring phenophases were stronger in the countryside compared to the city. In contrast, more pronounced urban trends were documented, for example by Defila (1999), Defila and Clot (2003) and Rötzer and Sachweh (1995) analysing long-term data of spring flowering dates in Germany and Switzerland. This is also in accordance with Primack et al. (2004) and Lavoie and Lachance (2006) who detected stronger trends in cities of North America and Canada using herbarium data. However, Jochner et al. (2012) did not find any urban effect on trends in phenological time series for nine spring phenophases in three German cities (see also Table 1). Whether the trends are stronger in urban or rural areas may be determined by the degree of urban development (Rötzer et al., 2000). Jeong et al. (2011) demonstrated that the magnitude of advancement in spring phenology is roughly proportional to the degree of urbanisation in South Korea. Therefore, the socio-economic background which influences urbanisation rates is a major factor in interpreting phenological time series and differences in urbanisation rates should be regarded in further studies.

The question whether first arrival dates of migration birds advanced faster in urban or rural areas was addressed by Tryjanowski et al. (2013). The authors found stronger trends for urban areas; thus, arrival dates in cities are fast approaching, or have now matched or passed those in the countryside.

3. Questions on the methodology

3.1. Space-for-time substitution

Information on changes in phenological responses to global warming can be obtained from long time series, experiments or natural climate gradients (Dunne et al., 2004). Highly heterogeneous city structures and a general decrease in urbanisation from the city centre towards the countryside provide a variety of temperature conditions (Jochner et al., 2013b). Concurrently urbanisation gradients eliminate the confounding effect of varying day length (Mäkelä, 2013), whereby spatial patterns can be translated into temporal patterns. This approach, the so-called space-for-time (SFT) substitution, is commonly used in ecology and forest science (Fukami and Wardle, 2005). In phenological research the SFT substitution might be valuable if the phenological response to climate change over time is equal to the response to climate variability over space (Dunne et al., 2004).

Jochner et al. (2013b) found that the temperature response rate (the advancement of a phenophase in response to an increase in air temperature by 1 °C) derived from spatial birch flowering data in Munich and Ingolstadt ($-4.4 \text{ days}/^{\circ}\text{C}$) was significantly different from that of long-term data in Bavaria ($-1.9 \text{ days}/^{\circ}\text{C}$). Local site effects such as differences in soil conditions, pollution, nutrient availability, etc., might lead to differences in temperature response rates. In addition, long-term studies are typically characterised by observing the same individual year after year. Thus, these species are subject to ageing which might influence the relationship between temperature and phenology (Rosenzweig et al., 2008). On the other hand, spatial data are based on numerous trees with different individual genetics that could imply differences in responses (Baumgartner, 1952). In addition, the relationship between phenological onset dates and temperature is usually considered as linear, however, a sigmoidal (s-shaped) relationship with upper and lower limits at the temperature extremes is expected to be more realistic (Sparks et al., 2000). Thus, temperatures close to the extremes of a species tolerance might elicit a less pronounced response. Moreover, tree acclimation to higher temperatures could result in a progressive reduction of temperature response rates (Eccel et al., 2009).

Buyantuyev et al. (2012) noted that some plant phenological records were either short or had missing data implying uncertainties in the interpretation of the direction and magnitude of change. To overcome these limitations and to fill gaps in historical records of ornamental tree species in Hohhot, China, the authors used spatial phenological and temperature data recorded in the same urban landscape. In the case of substantial incomplete historical records or small sample sizes, they were able to increase the robustness of phenological models. However, they also found that the majority of estimates of historical trends were not noticeably affected. In addition, the variation of spatial data was consistently lower; thus spatial data cannot ultimately be used to substitute for historical records.

A contrary finding was presented by Youngsteadt et al. (2014) who reported that biological responses to urbanisation and global change were similar. The authors analysed a various set of abundance data of the herbivore gloomy scale (*Melanaspis tenebricosa*) including data derived from urban and rural leaf samples as well as historical herbarium samples of red maple (*Acer rubrum*). They found that maximum insect abundance in historical and urban

datasets occurred at similar temperatures. However, they also detected stronger temperature responses in the city than in historical rural herbarium samples and proposed that factors such as CO₂ and nitrogen deposition and their interactions with temperature may be responsible for these differences.

Buyantuyev et al. (2012) pointed out that there are still unexplored opportunities in bringing temporal and spatial perspectives together. Incorporating long-term and spatial data would be helpful since some statistical procedures require a huge sample size, e.g., locally weighted scatterplot (LOWESS) regression which is used to investigate non-linear relationships. They also warned that climate-based phenological models assume that phenological responses to environmental factors are stable through time and across space (Pau et al., 2011) but the validity of these assumptions remains to be tested.

High urban temperatures might be useful to predict future conditions for which past observations offer no comparison (Wolkovich et al., 2012). Therefore, it is also essential to extend the number of observation years within spatial studies since a variety of different weather conditions will extend temperature variations. Megacities, which were largely neglected in phenological studies, may also show a greater UHI effect leading to greater urban–rural temperature differences and deeper insights in future changes.

3.2. From “black and white” to a spatial coverage

Not only the study method but also the study design is of importance in urban phenology. Research is often restricted to a limited number of observation sites within cities in existing phenological networks. Several studies incorporated only stations that were situated outside the urban core (Rötzer and Sachweh, 1995; Koch, 1986) and most of the research was based on a simple urban–rural dichotomy (Rötzer et al., 2000; Gazal et al., 2008; Mimet et al., 2009). This can produce unexplainable and inconsistent results of differences in plant phenology, especially when the city as a whole is compared with rural sites and any intra-urban variation is ignored. Yet, many authors (e.g., Franken, 1955; Baumgartner et al., 1984; Bernhofer, 1991) reported substantial phenological differences within the city. Bernhofer (1991), for example, showed that forsythia (*Forsythia suspensa*) flowered 20 days earlier in the inner city of Vienna, Austria, compared to an urban park; Fotiou et al. (2011) found differences in flowering of 12 days between a south- and a north-facing population of wall-pellitory (*Parietaria judaica*) in Thessaloniki, Greece. Furthermore, a pure “black and white” analysis may be limited since the UHI effect extends beyond the city, whereas a substantial influence on phenology can be observed up to 6 (Han and Xu, 2013) and 10 km (Zhang et al., 2004) beyond the edge of urban land cover. Urban land cover was even found to influence spring onset and autumn offset in the US mid-Atlantic region to a distance of 32 km from large cities (Elmore et al., 2012).

Several authors suggested that the distance from the urban core may be a good predictor of phenological changes associated with urbanisation. The relationship between phenological onset dates and distance to the city centre was regarded as a linear function by Mimet et al. (2009). In contrast, Fisher et al. (2006) found a power relationship between distance to the urban core of Providence (Rhode Island) and satellite derived onset dates. Furthermore, Zhang et al. (2004) presented an exponential fit between onset dates derived from satellite data and distance to the urban core. Han and Xu (2013) found that the effect of urbanisation on land surface phenology was strongest near urban areas and decreased logarithmically with the distance to the urban edge.

Commonly, satellite derived land cover classifications are utilised to achieve a categorisation of urban and rural sites (White

et al., 2002; Gazal et al., 2008). In addition, present day and historical topographic maps (Rötzer et al., 2000), MODIS NBAR spectral data, night time light data and gridded population density (Zhang et al., 2004) can be used. A classification of urban, suburban and periurban sites for the city of Rennes (France) was performed by Pellissier et al. (2008) using digital orthophotos within 500 m × 500 m grid squares to calculate the proportion of various land use classes (surface water, woodland, cropland, grassland, roads, built-up areas) by means of principal components analysis (PCA). This categorisation in three different levels of urbanisation was also implemented by Mimet et al. (2009). A further estimate of the degree of urbanisation was achieved by Shustack et al. (2009) who used orthophotos to evaluate the proportion of different cover types (forest, agriculture, mowed, paved, road) and the number of buildings within a radius of 1 km using PCA. The first component of the PCA was defined as the index of urbanisation, where positive and negative values indicated urban and rural sites, respectively.

Rötzer et al. (1997) calculated a building index based on digital land use information that comprised a summation of raster elements with densely developed areas within a radius of 2.5 km. Jochner et al. (2012) developed an index reflecting the degree of urbanisation using CORINE land use data and GIS. This index, from zero to one, indicates the percentage of predefined built up areas (e.g., continuous and discontinuous urban fabric, industrial or commercial units) within a radius of 2 km. Jochner et al. (2013a) denoted urban sites as those with at least 50% urban cover (urban index ≥ 0.5) and the remainder as rural sites. Comber and Brunsdon (2014) used data from the Land Utilization Survey of Britain which includes urban and suburban classes, as well as the Land Cover Map 2007 for obtaining information on the proportion of built-up areas and gardens at a 1 km level. This information was used to calculate the proportion of urban land use in 10-km grid squares.

Although the analysis of urban–rural gradients, especially when long time series are available, provides deep insights in the spatial variation of phenological onset dates only satellite derived metrics provide a comprehensive spatial coverage. However, these data, as already mentioned (Section 2.1), only refer to the ecosystem level (Walther et al., 2002). Series of satellite derived metrics are comparably short and the date of spring onset can vary substantially, depending on the algorithm applied (White et al., 2009). These inherent methodological differences imply that there is no strong association between phenological onset dates derived from field observations and remote sensing data (Badeck et al., 2004; Cleland et al., 2007). Thus, there is still a need for comparison studies combining both *in situ* observations and NDVI-based satellite data (Fu et al., 2014). Nevertheless, satellite derived data was found to adequately depict the magnitude and form of urban heat island effects on vegetation phenology (Zhang et al., 2004). For example, Zhang et al. (2004) reported that vegetation greenup in eastern North America was advanced by 3 days per 1 °C increase in temperature – a result which is comparable to values obtained from field observations (e.g., Rötzer and Chmielewski, 2001). Further research in urban environments should focus on a multi-scale approach combining observational and satellite derived data in order to compare the results derived from these different methods and to prevent an over- or underestimation of phenological onset dates in cities.

4. Implications of urban–rural differences in phenology

4.1. Feedback on vegetation

Phenological changes might lead to shifts or changes in other ecological processes (Luo et al., 2007), for example, related to the

timing of ecosystem services availability to urban inhabitants (Jenerette et al., 2011). Ecological mismatches are the results of dissimilarities in phenological changes for different species. Plant phenology is closely related to that of insects; they, in turn, are a major food source for birds (Visser et al., 1998). A prominent example highlighting the implications of climate change on species interaction is the pied flycatcher (*Ficedula hypoleuca*). This migratory bird arrives in northern Europe each year in spring, but now misses the peak of insect abundance; hence, the feeding of its nestlings became problematic (Both and Visser, 2001). In addition, the availability of nest sites for breeding birds is also influenced by plant phenology (Hansell, 2000). Delayed (Tryjanowski et al., 2013) or advanced (Miller-Rushing et al., 2008) arrival dates of birds in urban areas may disrupt intrinsic interactions and processes and may affect bird survival and reproduction (Primack et al., 2009). Divergences in flowering times between urban and rural areas may also lead to reproductive isolation; especially with respect to plants that have a short flowering duration. A condensed or missing overlap of the flowering period of insect-pollinated plants may impact the sharing of pollinators and the time of pollen availability (Primack et al., 2009). Different responses of plant phenology between urban and rural environments could block or restrict gene flow among meta-population and meta-community in rural–urban transects (Neil and Wu, 2006). In addition, these different responses are likely to accelerate species polarisation (Neil and Wu, 2006). However, detailed investigations on the impact of urban temperatures on the timing of ecosystem services availability and resulting ecological mismatches are rare. This firstly arises due to a lack of studies which specifically address plant trait induced differences in urban areas and, secondly, because of a deficiency of studies investigating the phenology of birds, insects and other animals in cities.

Earlier phenological onset dates in spring and a longer vegetation period may result in an increase of frost damage (Walther et al., 2002). This is especially important for productivity and yield of agriculture and horticulture. In urban areas of Phoenix, USA, higher urban temperatures along with a low risk of frost allow oranges and Bougainvilleas to grow (Baker et al., 2002). However, heat stress contributes to a lower quality of local cotton crop implying a decreasing profitability for agriculture at the urban fringe (Baker et al., 2002).

4.2. Feedbacks on climate

In general, a prolongation of the vegetation period is associated with changes in biogeochemical processes such as photosynthesis or carbon sequestration and physical properties of vegetated land surfaces related to, for example, surface energy or water balance (Peñuelas et al., 2009).

The assessment of the length of the vegetation period is particularly important in urban areas since an extended growing season is linked to a longer time for plants to be active which likely involves an increase in annual CO₂ uptake and a mitigation of global warming (Keeling et al., 1996; Peñuelas and Filella, 2001; Peñuelas et al., 2009). However, White and Nemani (2003) found that a longer canopy duration was only associated with an increase in carbon sequestration when the precipitation of the warm season was above average.

In addition, a positive effect associated with longer leaf seasons is the effective deposition of ozone (O₃) and particulate matter (Nowak et al., 2000; McDonald et al., 2007). This, however, counteracts the emission of biogenic volatile organic compounds (BVOC) which do not only serve as precursors of O₃ (Cortinovis et al., 2005; Simpson and McPherson, 2011) but are also involved in aerosol formation (Peñuelas and Llusia, 2003; Peñuelas et al., 2009).

Additionally, phenology influences meteorology on the meso-scale via partitioning net radiation into sensible and latent fluxes (Fitzjarrald et al., 2001; Schwartz and Crawford, 2001). A prolongation might not only result in changes in evapotranspiration but also in changes related to turbulence and albedo, processes which also affect near-surface climate (Peñuelas et al., 2009; Jeong et al., 2011). Comprehensive investigations about meteorological conditions altered by differences in phenology are, however, lacking. Peñuelas et al. (2009) warned that the prolongation of the vegetation period can – depending on water availability and regional characteristics – either reduce or increase temperature: additional water vapour in wet regions/seasons is associated with increased cloud formation resulting in surface cooling and increased precipitation. Vegetation in drier regions/seasons might be affected by absorbing more sunlight but not with markedly amplified evapotranspiration.

4.3. Allergy and human health

Climate change related effects on aeroallergens have already been observed in terms of pollen amount, pollen allergenicity, timing and duration of the pollen season, plant and pollen distribution and episodes of long-range transport (e.g., reviewed by Beggs, 2004; D'Amato and Cecchi, 2008).

The impact of urban temperatures on the phenology of some allergenic species has already been analysed (e.g., Ziska et al., 2003; Jochner et al., 2011), however, detailed studies on allergenic vs. non-allergenic plants have not yet been undertaken. Allergenic plants which are mostly wind-pollinated are among the earliest species to flower in the year (Ziello et al., 2012a). Since phenophases that occur early are especially sensitive to temperature (Menzel et al., 2006), past and future warming had, and will have, significant effects on flowering onset dates of allergenic plants. For example, the footprint of cities on flowering dates of allergenic plants has been demonstrated to be especially high for common hazel (*Corylus avellana*), an important allergenic plant species in Europe (Jochner et al., 2012).

Urban–rural differences in the start and the duration of the flowering period of allergenic plants may result in substantial disorders in terms of human health. An earlier urban onset would imply that the pollen season also starts earlier in the city compared to the countryside. Since pollen grains are transported through the air in more or less discrete clouds and considerable time and distance are required before a homogeneous distribution is reached (Spieksma, 1980) pollen allergic people might benefit from rural pollen-free conditions. Urban–rural differences in phenology are not universal across years; thus, Jochner et al. (2011) suggested that pronounced urban–rural differences in phenological onset dates (associated with cold spells in spring) might lead to a longer time period during which pollen allergic people experience symptoms (due to a steady development and transport of pollen in the atmosphere from the urban to the rural settlements and vice versa). Warm spells which were shown to lead to a simultaneous flowering in urban and rural environments, in turn, do not imply a delayed pollen exposure in the countryside. Instead, the number of plants flowering at the same time might involve a greater abundance of airborne pollen and therefore stronger symptoms in pollen allergic people. In general, interrupted pollen seasons and therefore perpetual minor and major ailments could be the result of very early flowering onset dates, since weather in late winter and early spring are often connected with adverse (colder and more humid) conditions (D'Amato and Cecchi, 2008). Flowering periods of different allergenic species might also overlap (Doi et al., 2010) and could result in a reduction of the allergy-free time and hence in reinforced ailments for pollen allergic people.

Studies in urban environments are not only informative about differences in phenological onset dates, but also about other plant characteristics. Ziska et al. (2003) found that pollen production of common ragweed increased along an urban–rural gradient in Baltimore, USA. However, there are also studies identifying adverse environmental conditions in cities which might probably restrict pollen production of white goosefoot (*Chenopodium alba*) in Porto, Portugal (Guedes et al., 2009) and birch in Munich (Jochner et al., 2013c). In contrast, Ziello et al. (2012b) found that pollen production of various plant species across Europe was increased, likely due to global change related increases in atmospheric CO₂ concentrations. This indicates that there is still a lack of knowledge concerning pollen production under different environmental conditions. Contrasting results were also provided concerning the allergenicity of pollen specimen in cities: Ziska et al. (2003) and Guedes et al. (2009) reported decreased allergen content of common ragweed and white goosefoot, respectively. The major birch allergen Bet v 1 was found to be enhanced at rural sites of Munich, probably a fact related to amplified rural O₃ concentrations (Beck et al., 2013). Since existing studies have reported an increase of Bet v 1 levels under elevated temperature conditions (Ahlholm et al., 1998; Hjelmroos et al., 1995) there might indeed be other influential factors (e.g., pollutants) which are responsible for differences in allergenicity found in these studies. Thus, further investigations on the allergenicity of pollen specimen are urgently needed. Urban areas do not only offer preview of future pollen related problems (Ziska et al., 2003). Several studies have demonstrated that urban dwellers (especially children) are predominately influenced by pollinosis incidence (e.g., Riedler et al., 2000; Bibi et al., 2002), probably due to a westernised lifestyle, urbanisation-induced temperature increase and abundance of air pollutants (Braun-Fahrlander et al., 1999; D'Amato, 2000; Ring et al., 2001). This again emphasizes the importance of cities and the need to further focus on research conducted in urban environments.

5. Conclusions

This review paper summarising most of the current literature in the field shows that urban phenology can be used for the estimation of the UHI effect. However, several confounding factors should be taken in consideration which weaken the relationship between phenology and temperature and modify urban–rural differences in phenology: Extreme weather conditions in spring might lead to more or less pronounced variations in onset dates; for example, nutrients, precipitation and edaphic factors might also impact urban phenology. These influences might be species-specific and vary across geographical regions. In addition, this review elucidated that urban phenology is useful as an empirical proxy for climate change related impacts on phenology. However, differences in the study design of spatial and long-term analyses hinder the generalisation of one specific study method. The implications of urban–rural variations in onset dates are associated with various impacts on vegetation (e.g., timing of ecosystem services, ecological mismatches or frost damage and heat stress), meteorology (e.g., carbon sequestration, surface energy or water balance) and human health (e.g., earlier start and longer duration of the pollen season). However, a deeper understanding on the consequences of advanced urban onset dates is further required. In general, there are a number of questions that still need to be answered:

- Are urban–rural differences related to pollination types?
- What is the impact of the UHI effect on autumn phenology?
- Can satellite derived data in Europe be used to estimate the length of the vegetation period in urban areas?

- What are the urban–rural differences in phenological events of animals, for example, birds?
- How do extreme temperatures affect urban phenology?
- What are the effects of light pollution on the phenology of urban species?
- Are urban temperatures in winter still low enough to fulfil chilling requirements?
- Is urban soil temperature or urban–rural differences in soil temperature useful for phenological modelling?
- What is the influence of nutrients, assessed by soil and foliar analyses, on phenology?
- How does groundwater depression or whole metropolitan water fluxes (especially in arid regions) influence urban phenology?
- Which species are adequate for the detection of the UHI effect in the tropics?
- What further factors influence tropical phenology?
- Can differences in urban and rural temporal trends in phenology be explained by socio-economic factors?
- What is necessary to bring temporal and spatial phenological data together?
- Do more (extreme) observation years and observations from megacities improve phenological estimates due to sharper temperature gradients?
- Are studies which incorporate both satellite derived and observational phenological data in urban environments comparable?
- What are the consequences of urban–rural differences in the timing of ecosystem services availability?
- What important potential ecological mismatches are related to these urban–rural differences?
- How are meteorological conditions altered by urban–rural and intra-urban differences in phenology?
- Are the effects of the UHI more/less pronounced for allergenic vs. non-allergenic plants?
- How does urban environmental condition alter pollen production and allergenicity of allergenic species?

In general, urban phenological studies have moved from a rather descriptive analysis of urban–rural differences in onset dates or trends to a causal explanation of influential factors and the prediction of future changes. The great challenge will be to link data sampled at urban sites with existing long-term data. The inherent methodological differences of different study types should be considered and their individual advantages should be used to improve phenological models, resulting in a deeper understanding of ecological processes and changes associated with climate change.

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