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Ecological monitoring with citizen science: the design and implementation of schemes for recording plants in Britain and Ireland

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Interest in citizen science has been increasing rapidly, although the reviews available to date have not clearly outlined the links between the long-established practice of recording plant species' distributions for local and national atlases, or other recording projects, and the gradual development of more structured monitoring schemes that also rely on volunteer effort. We provide a review of volunteer-based plant monitoring in Britain and Ireland, with a particular focus on the contributions of expert volunteers working with biological recording schemes and natural history societies; in particular, we highlight projects and practices that have improved the quality of data collected. Although the monitoring of plant distributions at larger scales has led to numerous insights into floristic change and its causes, these activities have also led to the recognition that knowledge of species' abundances at finer-scales often provides a more powerful means of detecting and interpreting change. In the UK, this has led to the development of a new, abundance-based 'National Plant Monitoring Scheme'. We outline this new structured scheme, and review some of the design considerations that have been made during its development. New monitoring projects require a clear justification, and the launch of a new scheme is also an opportune moment to review whether some basic assumptions about the collection of monitoring data can withstand scrutiny. A distinction is often made between monitoring that is focused on answering particular, focused questions, and that which is more generally seeking to detect changes; for example, in species' distributions or abundances. Therefore, we also review the justification for such general 'surveillance' approaches to the monitoring of biodiversity, and place this in the context of volunteer-based initiatives. We conclude that data collected by biological recorders working within atlas or monitoring scheme frameworks will continue to produce datasets that are highly valued by governments, scientists, and the volunteers themselves. © 2015 The Linnean Society of London, *Biological Journal of the Linnean Society*, 2015, **115**: 505–521.

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INTRODUCTION

Long-term ecological monitoring schemes with volunteer participants are a large part of the landscape of what is now often referred to as 'citizen science'; that is, the participation of people who are not professional scientists in activities that contribute to scientific research (Dickinson, Zuckerberg & Bonter, 2010; Dickinson & Bonney, 2012). Although such initiatives may be increasingly linked to technological advances (August *et al.*, 2015), technology is not an essential component of citizen science as now understood. In many cases, existing schemes, or historical

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short-term initiatives, considerably pre-date the popularization of this term in the 1990s (Bonney, Shirk & Phillips, 2013). Indeed, despite some assertions that, up to the mid-1990s, 'volunteer-led data collection efforts were relatively few in number' and that 'most ... focused on monitoring the quality of lakes, streams and rivers' (Bonney *et al.*, 2013), data collection conducted largely by volunteers across large areas has been a standard part of natural history in Britain and Ireland since at least the 1962 *Atlas of the British Flora* (Harding & Sheail, 1992; Preston, Croft & Pearman, 2002b; Preston, 2013). This, in turn, has its roots in the network of mainly amateur botanists who assisted in the elucidation of the biogeography of the British and Irish flora in the 19th and early 20th Centuries (Allen, 1986). Its history in Britain and Ireland can be traced even further back, through to the huge contributions made by the earlier naturalists who laid the foundations for our understanding of the taxonomy and composition of our flora and fauna (Ray, 1660, 1670; Allen, 1976; Oswald & Preston, 2011). These types of historic observations still have great value today for helping us to understand long-term environmental change (e.g. Preston, 2000, 2003; Walker, 2003a, b; Walker, Preston & Boon, 2009), and it is now increasingly recognized that biological recording has played a pioneering role in involving nonprofessionals in the collection of scientific data (Pocock *et al.*, 2015). Other volunteer-based recording projects with early origins in Britain and Ireland include the UK Butterfly Monitoring Scheme (UKBMS), begun in 1976 (Pollard & Yates, 1993), and the British Trust for Ornithology (BTO)/Joint Nature Conservation Committee/Royal Society for the Protection of Birds Breeding Bird Survey (BBS), which has its origins in the Common Bird Census begun in the early 1960s (Harris *et al.*, 2014).

The term 'monitoring scheme' can refer to a wide range of activities, conducted at different scales, by amateurs or professionals, focusing on many different aspects of the environment (Spellerberg, 2005; Lindenmayer & Likens, 2010b; Gitzen *et al.*, 2012; Pocock *et al.*, 2015b). The production of species' distribution atlases can itself be considered as a form of monitoring, particularly where data can be assigned to different time periods (Telfer, Preston & Rothery, 2002; Spellerberg, 2005; Tulloch *et al.*, 2013), although atlas initiatives can typically only report on distributional change. Atlas (sometimes also called 'cross-sectional') projects, although potentially extremely valuable with respect to collecting information on the distributions and large-scale trends shown by species (Spellerberg, 2005; Robertson, Cumming & Erasmus, 2010; Porter & Leach, 2013; Hill & Preston, 2014; Stroh *et al.*, 2014), are not typically run on short time-scales according to tightly controlled data collection protocols (Brotons, Herrando & Pla, 2007). By contrast, monitoring schemes that allow the estimation of time trends in the abundances of animals or plants that can be regularly updated generally provide a more sensitive measure of change for the reporting demands of national or regional governments (Jones *et al.*, 2011). From this point on, we reserve the term 'monitoring scheme' for these 'BBS'-type programmes, at the same time not forgetting that other types of monitoring are possible. Existing British and Irish monitoring schemes, then, include the UKBMS and UK BBS previously mentioned, as well as the National Bat Monitoring Programme (Barlow *et al.*, 2015), Wetland Bird Survey (Austin *et al.*, 2014), and Wider Countryside Butterfly Survey (Brereton *et al.*, 2011). Many of these volunteer-based schemes have comparable programmes in other European countries (Schmeller *et al.*, 2009; see also http://www.floron.nl for current Dutch initiatives). National monitoring schemes that combine information from professional and volunteer surveyors also exist, such as in Switzerland (Pearman & Weber, 2007; FOEN, 2014). Monitoring schemes that are still in the developmental or scoping stages in the UK include the National Plant Monitoring Scheme (Walker *et al.*, 2010a, 2015) and a monitoring scheme for pollinators and pollination services to crops, although we note that volunteer-based, distribution-focused biological recording activities for these taxa (although not on pollination *per se*) have existed in Britain and Ireland for many years (Harding & Sheail, 1992; Spellerberg, 2005; Pocock *et al.*, 2015).

It is also important, however, to remember that the distinction between an atlas project and a monitoring scheme may often be blurred. Information on the presence of species at larger spatial scales can normally be derived from finer-scale monitoring scheme projects, whereas abundance data can also now be linked to larger-scale detection/nondetection data collected as part of an atlas project (Pagel *et al.*, 2014). In addition, a wide variety of techniques can also now be used to extract estimates of trends from biological records data that may be relatively unstructured *en masse*, or semi-structured where atlas-focused recording schemes encourage the use of a particular protocol (Isaac *et al.*, 2014). Indeed, the use of recording protocols and support networks for atlas projects (Dines, 1996; Walker *et al.*, 2010b; Balmer, Gillings & Caffrey, 2013) can help to break down the distinction between monitoring schemes and atlas-focused fieldwork, where the end result is higher-quality, more structured data. Similar suggestions, based largely on the premise of incorporating elements of monitoring schemes into the design of atlas projects, were recently made by Tulloch *et al.* (2013). These suggestions included fine-scale data collection; replication over time covering a range of habitats; and communication of data needs to volunteers. Regional coordinators for the curation of data and quality checking were also highlighted as desirable. Most of these ideas have been appreciated by many British and Irish biological recording schemes for some time (Preston *et al.*, 2002b; Pocock *et al.*, 2015a,b). The suggestions of Tulloch *et al.* (2013) were motivated by the finding of the greater scientific impact of monitoring scheme data. Regardless of whether one accepts the premise that the impact of a national, often largely volunteer-based piece of work, such as an atlas, can be adequately quantified by scientific citation counting, the introduction of more structure to volunteer-based survey work is clearly very important for data quality and, consequently, for the robustness of any ecological conclusions drawn (Walker *et al.*, 2010b).

Here, we (1) review volunteer-based plant monitoring in the broad sense in Britain and Ireland, particularly focusing on its contributions to our knowledge of ecological change during the late 20th Century; (2) outline approaches towards an abundance-based plant monitoring scheme; and (3) review the justification for such general 'surveillance' approaches to the monitoring of biodiversity.

VOLUNTEER-BASED PLANT MONITORING PROJECTS IN BRITAIN AND IRELAND

Where the recording of plant distributions and abundances are concerned, the Botanical Society of Britain and Ireland (BSBI), in collaboration with the national Biological Records Centre of the UK, has a long history of learning from the challenges posed by volunteer-based projects covering large areas (Rich & Woodruff, 1990; Perring, 1992; Croft & Preston, 1999; Spellerberg, 2005; Braithwaite, Ellis & Preston, 2006; Preston, 2013; Pocock *et al.*, 2015). For example, the *Atlas of the British Flora* (Perring & Walters, 1962) started life as a descriptive, phytogeographical project (Perring, 1992), although it was subsequently used to identify widespread declines in various species; these insights have come to be widely considered as one of the most significant applications of the data collected (Rich & Woodruff, 1990; Porter & Leach, 2013; Preston, 2013), and led to the first Red Data book of British plants (Perring & Farrell, 1977). The uses to which the *Atlas of the British Flora* data were put clearly highlight the potential value of distribution mapping for conservation. This particular case can be seen partly as the product of a particular historical and ecological situation: the periods across which the 1962 *Atlas* data were compiled were a time of great change in the British countryside (Perring, 1970), and particular habitats were being lost at an unprecedented rate (Fuller, 1987; Hooftman & Bullock, 2012). This meant that the relatively coarse scale of the *Atlas* (10 × 10 km) was able to reveal large changes in the distributions of many plants (Perring, 1970); these types of data are still regularly used to produce conservation assessments at a national scale (Stroh *et al.*, 2014; Van Maes *et al.*, 2015).

The far-reaching legacy of the 1962 *Atlas* (Preston *et al.*, 2002b; Preston, 2013) led to discussion during the 1980s concerning the need for an updated atlas, aiming to provide further information on plant distributions for both conservation and statutory demands (Rich & Woodruff, 1990). A report of the time notes the general feeling in the plant recording community that a new atlas survey during the late 1980s might coincide with a slowing of agricultural intensification, and that it might therefore be better to wait for a period of 'relative stability' to maximize the longevity and relevance of the maps created (Rich & Woodruff, 1990). This highlights one tension for co-ordinators of atlas projects: the snapshot captured by surveyors may miss a period of widespread change, leading to a resource with the potential to mislead its users. This may be one reason why some studies report that monitoring schemes appear to be more influential than atlases in the scientific literature (Tulloch *et al.*, 2013). In the case of the BSBI, the decision to resurvey for a new national plant atlas was not taken lightly: Rich & Woodruff (1990) note that there was much discussion in the 1980s about the appetite for a new volunteer survey. These doubts, coupled with worries that the distribution of many species might not have changed much since 1962, led to the planning of a different type of survey, namely the 'BSBI Monitoring Scheme' (Rich & Woodruff, 1990, 1996; Preston *et al.*, 2002b). This was designed not only to collect information at a finer scale (2 × 2 km) than the 1962 *Atlas*, using a regular, systematic grid sample, but also to allow comparisons to be made with the *Atlas*. The scheme was therefore intended both as a first step towards a new national plant atlas, and as the basis of a new type of scheme that would be run periodically: one that integrated detection/nondetection type atlas data with a rigorous sampling framework (Rich & Woodruff, 1990). This demonstrates that the BSBI Monitoring Scheme coordinators were identifying multiple uses for their future dataset, as well as considering the characteristics required for a scheme to provide answers to questions about changes in plant distributions, and providing important insights into the capacity for future volunteer-based atlas surveying (Rich & Woodruff, 1996). Moreover, between one-quarter and one-eighth of species investigated were found to have undergone significant change, depending on the region, despite the avowed lack of sensitivity of the project by the coordinators (Rich & Woodruff, 1996). Five general trends were also highlighted, including widespread losses of grassland taxa and large declines in arable weeds, which were later confirmed during analyses of the subsequent national *New Atlas* (Preston, Pearman & Dines, 2002c; Preston *et al.*, 2002a, 2003). This demonstrates that the 1987–1988 BSBI Monitoring Scheme was successful in detecting at least the larger changes that had occurred in the landscape since the completion of the 1962 *Atlas* (Preston *et al.*, 2002b).

To our knowledge, the BSBI Monitoring Scheme is the only example of a volunteer-based, systematic monitoring scheme for plants that has sampled such a large area (Britain and Ireland, an area of 315 130 km^2; Rich & Woodruff, 1996), and it must certainly be the earliest such scheme successfully completed. However, the BSBI Monitoring Scheme appears to have been largely overlooked by the wider ecological monitoring and citizen science literatures; for example, of the 110 papers indexed by Google Scholar as citing Rich & Woodruff (1996) as of December 2014, none are general reviews of volunteer-based ecological monitoring or atlas projects, despite reviews of this area being published with increasing frequency (Cohn, 2008; Bonney *et al.*, 2009; Silvertown, 2009; Devictor, Whittaker & Beltrame, 2010; Dickinson *et al.*, 2010, 2012; Robertson *et al.*, 2010; Hochachka, Fink & Zuckerberg, 2012; Miller-Rushing, Primack & Bonney, 2012; Tulloch *et al.*, 2013). This neglect may be partially because the botanical findings were only reported in two scientific papers (Rich & Woodruff, 1996; Rich, Beesley & Goodwillie, 2001), with other commentaries on the findings being published largely in the grey literature (Rich & Woodruff, 1990; Palmer & Bratton, 1995; Rich, 1996; but see also Pearman *et al.*, 1998). The project coordinators also emphasized the possible biases in the data collected (Rich & Woodruff, 1992; Rich, 1998), demonstrating the desire to ensure that the findings were as robust as possible. This lead to further work on systematic sampling for volunteer-based plant surveying (Rich & Smith, 1996), resulting in an attempt at recording a local flora with uniform effort (Rich *et al.*, 1996). In this sense, the Monitoring Scheme coordinators were amongst the earlier innovators in an area now receiving much more attention; namely that of dealing with bias in distribution data (Dickinson *et al.*, 2010; Robertson *et al.*, 2010; Isaac *et al.*, 2014). The BSBI Monitoring Scheme was not a one-off, and the initial intention to carry out repeat surveys was fulfilled in 2003–2004 when the BSBI follow-up project 'Local Change' was carried out in Britain using the same methodology and grid squares as those of the earlier scheme (Braithwaite *et al.*, 2006). This re-survey allowed the first analysis of national change in plant distributions at the 2 × 2 km scale. Seven-hundred and fifty volunteer recorders took part.

The Local Change survey was successful in creating distribution-based trend statistics for a large number (N = 725) of species. Species were also clustered into groupings, based on an *a priori* habitat classification and geographical distribution, for the purposes of creating more robust analyses of change (Braithwaite *et al.*, 2006). Considerable effort was also invested in adjusting for differences in recorder effort between the Monitoring Scheme and Local Change surveys (Braithwaite *et al.*, 2006, pp. 353–368). Local Change was highly successful in detecting signals of ecological change: it not only supported many of the earlier conclusions of the 2002 *New Atlas* (e.g. the loss of species of infertile habitats such as dwarf shrub heath; Preston *et al.*, 2002c; Braithwaite *et al.*, 2006), but also revealed new trends, such as the first evidence for a 'modest' recovery in the weed flora of arable fields (Braithwaite *et al.*, 2006: 323). However, as for the BSBI Monitoring Scheme, the Local Change project has not been widely cited as an example of a successful and innovative volunteer-based monitoring project: of the 70 citations of Braithwaite *et al.* (2006) on Google Scholar as of December 2014, none are from recent reviews covering the general areas of volunteer-based monitoring or citizen science. The authors of the BSBI Local Change report also reviewed the survey's contribution to UK plant monitoring and found that, in terms of species distribution types (e.g. rare, local, widespread, etc.) and habitat coverage, it filled a niche that was complementary to other existing professional and volunteer-based projects (Braithwaite *et al.*, 2006: 327). This complementarity is clearly an important consideration for projects aiming at longevity and efficiency, and the recognition of the unique contribution of the Local Change project has recently been re-affirmed by the BSBI, with a decision to re-run the survey in the early 2020s.

In addition to the innovations characterized by the 2 × 2 km, repeated, systematic surveys discussed above, traditional large-scale atlas data (i.e. 10 × 10 km) in broad date classes (Perring & Walters, 1962; Preston *et al.*, 2002b) are being increasingly used to draw inferences about trends. Methods of analyzing this type of data have recently been reviewed and evaluated by Isaac *et al.* (2014). Several of these techniques were developed in the context of botanical data (Telfer *et al.*, 2002; Hill, 2012) and are now being used increasingly on plant (Preston *et al.*, 2002c; Stroh *et al.*, 2014; Hill & Preston, 2014; Hill & Preston, 2015; Pescott *et al.*, 2015) and animal data sets (Balmer *et al.*, 2013; Cham *et al.*, 2014; Fox *et al.*, 2014). Other methods of analyzing floristic change, such as looking at changes in extrapolated occupancy probabilities using kriging (Groom, 2013) or other spatial methods (Le Duc, Hill & Sparks, 1992; Firbank *et al.*, 1998), have also been enhanced by the existence of the regular 2 × 2 km grid surveyed by the BSBI Monitoring Scheme and Local Change projects. It is clear from the applications seen in the ecological literature that, in Britain and Ireland, largely volunteer-collected atlas data have contributed an enormous amount to the national monitoring capacity (cf. Schmeller *et al.*, 2009) and to ecological research (e.g. McClean *et al.*, 2011; Powney *et al.*, 2014). Although at least partly recognized (Spellerberg, 2005), this fact has not received the attention that it deserves in the citizen science literature. Recent historical overviews of the subject have not clearly drawn out the continuous thread linking early biological recorders and taxonomists, such as John Ray, to current day recording activity (cf. Miller-Rushing *et al.*, 2012), have focused largely on birds (Dickinson *et al.*, 2010) or on North American contributions (Cohn, 2008; Miller-Rushing *et al.*, 2012). As noted by Pocock *et al.* (2015a) in the general context of biological recording, this has tended to mean that 'the distinctive attributes and successes' of British and Irish volunteer-based plant monitoring have been 'largely untold outside of the UK'; no doubt similar statements could be made about long-running schemes on the continent (Schmeller *et al.*, 2009). Nonetheless, all of the schemes discussed in the present review fit within standard definitions of ecological monitoring [e.g. Suter II (1993: 505) defines monitoring as the 'measurement of environmental characteristics over an extended period of time to determine status or trends in some aspect of environmental quality'] and they have also all been dependent on largely voluntary biological recording activity. The unique contributions of plant recording to the development of volunteer-based monitoring deserve to be more widely recognized (Preston, 2013).

It may be that atlas projects and related activities are seen as lacking the rigour of abundance-based monitoring schemes with more statistical designs (Tulloch *et al.*, 2013); however, it is inevitable that there will always be a desire, either from funders or from within a biological recording scheme or society itself, for interpretation and commentary on results from atlas or other cross-sectional projects, and, as has been emphasized above, many recording schemes already incorporate numerous 'monitoring scheme' elements of rigour in their make-up, resulting in data that tend towards being semi-structured rather than opportunistic. Regardless of commentaries in the academic literature, it is probably inevitable that atlas data will continue to be used to answer questions about environmental change: the emphasis on collecting and publishing atlas data within recording schemes attests to its popularity with amateur naturalists (Pocock *et al.*, 2015a). Indeed, in Britain and Ireland, the number of botanical county or regional atlases featuring distribution maps shows no signs of slowing (Fig. 1), and it is clear that the collected data have numerous potential uses (Hill, 2003). Analyses of change using atlas data also have the advantage of longer available time-series in many instances, albeit potentially with increased uncertainty around historical estimates of frequency.

DESIGNING NEW VOLUNTEER-BASED SURVEILLANCE MONITORING FOR PLANTS

In addition to the many uses of atlas-type plant distribution monitoring in Britain and Ireland, recently, and partly as a result of the Local Change survey (Braithwaite *et al.*, 2006), there has been growing recognition of the fact that observations of plant trends across finer scales of space and time would also be desirable, particularly where there is a wish

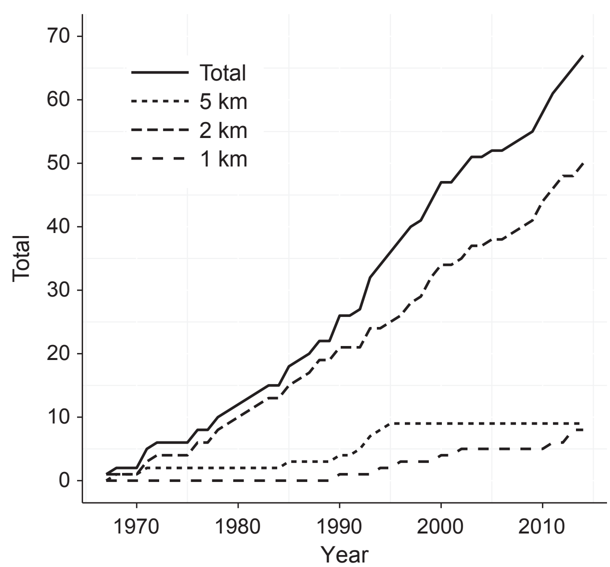

Figure 1. The cumulative number of plant distribution atlases published for British and Irish counties and regions at different grid sizes. The first local Flora featuring grid-based distribution maps was published in 1967 by J. G. Dony (1967).

to monitor population changes, or changes in species groups, at the scale of the habitat, or a desire to produce annual trends. Observations at this level are more likely to provide early warnings of negative trends or, indeed, of conservation successes. The need for a standardized approach to plant monitoring that would provide timely and robust estimates of status and trends was therefore identified as a high priority within the UK government's *Terrestrial Biodiversity Surveillance Strategy* (JNCC, 2008). A scoping study to investigate how this might be achieved soon followed (Walker *et al.*, 2010a). Walker *et al.* (2010a) reviewed a number of possible approaches to the deployment of volunteer-based plant monitoring at the habitat scale, drawing on existing schemes such as the BSBI Monitoring Scheme/Local Change, and approaches used in other species groups. Statistical design is obviously central to the robustness of any monitoring scheme, and, unless a project is solely aimed at public engagement, a defensible statistical foundation will be essential (Legg & Nagy, 2006; Jones, 2013). Below, a review is provided of the central requirements relating to the development of a nascent annual monitoring scheme for plants in the UK, namely, the new National Plant Monitoring Scheme (Walker *et al.*, 2015).

Scheme design

The power of a scheme to detect change

The statistical power to detect a specified level of change is often highlighted as a key preliminary for the design of any new monitoring scheme (Legg & Nagy, 2006), and much work has been carried out in recent years aiming to develop new models for estimating power that are appropriate to different types of monitoring data (e.g. Roy, Rothery & Brereton, 2007; Irvine & Rodhouse, 2010). Given the numerous sources of variance that may influence the ability of a large-scale, long-term, volunteer-based survey to detect change, a key challenge is to derive sensible parameters for power analyses. Variables that are typically explored include the number of years of monitoring; the number of sites; the number of visits within years; the effect size(s) to be detected; site revisit schedules; and variation affecting the trends exhibited across sites and species. For larger volunteer-based schemes, the choice of which variables to include in a prospective power analysis (and at what values) may be subject to considerable discussion. For example, for a scheme that intends to monitor aggregated trends across multiple species, how should trends simulated for power analyses be distributed across species? Power to detect an aggregated trend may be high if all species are assumed to follow a particular trajectory, whereas power may be low where species have randomly assigned trends. Similarly, for a single species across sites, the variance in a simulated trend will be a key determinant of power. Existing data are indispensable for estimating sensible variance parameters to investigate but, even where suitable data are available, key factors may still vary between the data utilized and the future reality of the scheme (Urquhart, 2012). For example, power analyses for the new UK National Plant Monitoring Scheme utilized long-term data from the Countryside Survey (http://www.countrysidesurvey.org.uk) to estimate variance parameters (O.L. Pescott, M. Jitlal and S.N. Freeman, unpubl. data), although Countryside Survey data are collected by professional ecologists, and may underestimate the amount of variation exhibited by a volunteer-based scheme. Of course, the 'true' variance in trends across sites may also change over time.

Site revisit schemes can also be important for power, particularly over relatively short time-scales. For example, sampling schemes that revisit sites in the medium-term generally have higher power that those that do not revisit sites, or only revisit them in the very short-term (Urquhart, 2012). Current recommendations highlight the need to revisit sites after a sufficiently long period for significant change to have occurred, but also to sample a large number of different sites. Sampling larger numbers of sites improves estimates of site variance, and leads to more precise estimates of trend sizes (Urquhart, 2012). One way of fulfilling these dual aims would be to have a rotating panel-like scheme for site revisits, supplemented with a set of sites that are visited each year (Urquhart, 2012). However, even with this intention, such a scheme is unlikely to be fully met in a volunteer context, due to surveyor drop-out, or differing preferences for visiting the same site every year versus visiting a number of sites on a rotating basis. Urquhart & Kincaid (1999), however, showed that random revisit schedules, with a mix of visits to new sites and revisits to existing sites, comes close to achieving the power of more systematic panel designs.

In general, recent approaches to power analyses using simulation-based estimates have increased the ease and flexibility with which the ecologist can estimate power (Gelman & Hill, 2007; Bolker, 2008; Irvine & Rodhouse, 2010). Analyses can now routinely use these approaches to include random effects to account for variation in responses (Miller & Mitchell, 2014; Johnson *et al.*, 2015). However, the incorporation of variance parameters in simulated data can sometimes yield misleading results. For example, the use of the Poisson distribution to simulate count data, combined with estimates of variances in trends from existing data, can lead to the counter-intuitive result that power increases with increasing variance; for example, for scenarios where initial counts and effect sizes (i.e. trends) are both small. Under these circumstances, simulated declines can lead to a small proportion of sites increasing to unrealistically large values. These outlying sites can then lead to the detection of significant increasing trends. The use of sophisticated models for estimating power clearly requires vigilance, as well as a constant need to check that simulated data are biologically realistic and that appropriate models are being used, particularly where investigated variances are derived from small existing datasets. Finally, a proliferation of modelled scenarios may actually make it harder to decide whether a study is likely to be under-powered or not, although it is also possible that bringing uncertainty around power analyses into the open may help to reduce unrealistic expectations.

Dealing with potential biases

Bias in monitoring schemes can arise in a number of ways, and may well be greatest for volunteer-based schemes. For example, organizers may find it challenging to persuade surveyors to visit random locations, particularly in an intensively managed landscape where the chances of encountering many target species may be low. However, specifically targeting nature reserves or other high-quality habitats leads to a biased picture of the countryside that is likely to be inappropriate for a scheme producing data to support mandated monitoring (Jones *et al.*, 2011). This was a potential criticism of the original UKBMS, in which participants selected their own survey locations, typically in areas of relatively high species richness for butterflies (Roy *et al.*, 2015). Butterfly Conservation, the Centre for Ecology & Hydrology, and BTO recently addressed this challenge by launching a new scheme, the Wider Countryside Butterfly Survey (WCBS), as a component of the UKBMS (Brereton *et al.*, 2011). The WCBS requires surveyors to visit a random (1×1 km) grid-cell at least twice a year. It predominately samples populations of common species, highlighting a trade-off between a representative design (i.e. a random sample of the countryside) versus the need to monitor rarer species that tend to be restricted to higher-quality semi-natural habitat (i.e. those targeted by the traditional transect location of the UKBMS). Initial results from the WCBS suggest that the trends are similar with either approach (Roy *et al.*, 2015). Given existing knowledge of the importance of high-quality habitat for many specialist butterflies (Warren *et al.*, 2001), and the fact that they are generally highly mobile organisms (Thomas, 2005), this result is not unexpected; however, without the WCBS, it would be impossible to know with any degree of confidence whether nature reserves were different from the wider countryside for commoner butterfly species, or to detect increases in the ranges or abundances of our rarer species occurring outside of the potentially biased set of traditional UKBMS sites.

One option for reducing bias from a set of sites, at the same time as retaining a focus on higher-quality sites that may lead to increased volunteer retention, is to create a weighted random selection, where the weights are derived from some measure of habitat quality. Existing land cover maps can be used to create weightings that integrate national frequencies of ‘high value’ habitat types with the areas of these types within a grid cell; the resulting weights can then be used in analyses, thus adjusting for the inherent bias (Yoccoz, Nichols & Boulinier, 2001). Although appealing, and often useful, this approach does have its limitations. Land cover maps are not perfect (Gerard *et al.*, 2012), and this approach will only be as successful as the quality of the land cover map used and the appropriateness of the land cover categories mapped for the monitoring question being addressed. Even if the bias in larger-site selection is accounted for in this way, bias can also arise from what occurs at the finer-scale of monitoring activity. For example, the placement of a transect or plot within a site may also lead to inaccurate assessments of a species’ status: surveys that are along rights of way are likely to overestimate the abundances of species of edges or other linear features, particularly for sessile organisms.

Surveys designed to detect both increases and decreases in some resource are also likely to be affected by surveyors self-selecting survey points: for example, the selection of the most typical example of a vegetation type is likely to mean that it contains most of the species that the surveyor is being asked to monitor; consequently, these patches are more likely to ‘decline’ in status on average (an example of the well-known phenomenon of ‘regression to the mean’). The challenge with all these types of bias is that the extent to which they may or may not affect results is essentially unknowable without conducting a second, unbiased, survey. The importance of getting it right first time can be appreciated. All of this argues for the unbiased allocation of survey points within a site, particularly for sessile organisms, although this is challenging for surveys by volunteers, particularly where much of a landscape is in private ownership. Obtaining unbiased trends, then, should be a key consideration of any monitoring scheme (Bart & Beyer, 2012), although, in cases where volunteers are contributing to a project, it is likely that bias of some sort or another cannot ever be entirely removed, except perhaps where projects focus on small areas. Techniques such as double-counting (i.e. two volunteers recording the same square independently), as is being implemented for the Dutch Butterfly Monitoring Scheme (C. van Swaay, pers. comm.), combined with occupancy modelling may help to alleviate certain forms of potential bias (Chen *et al.*, 2009).

Selecting the species to monitor

It is well established that monitoring schemes often focus on subsets of species as a result of resource constraints (Manley *et al.*, 2004) or to enable wider participation in a scheme. Even where experienced volunteers are encouraged to record all species encountered in a group, low encounter rates for rare species may still restrict the number of species for which trends can be produced (Braithwaite *et al.*, 2006). For the new UK National Plant Monitoring Scheme, a particular focus has been on encouraging broad participation. This has led to a scheme design that attempts to appeal to different skill levels: an entry level, composed of short lists of around 15 plants per habitat; an intermediate level, with lists of up to 30 species per habitat, including harder to identity groups such as grasses, sedges, and ferns; and an expert level, where all species within a plot are recorded. With 28 habitats included in the scheme (Walker *et al.*, 2015), this means that the entry and intermediate levels contain approximately 200 and 400 species, respectively. One UK public engagement scheme, 'Wildflowers Count', led by the conservation charity Plantlife, found that slightly shorter overall lists (99 species, in this case) were popular with volunteers, although many participants asked for more species aiming to increase the number of encounters in the field. A list of 400 species for an intermediate level of volunteer expertise may appear excessive, although it should be remembered that a volunteer is unlikely to encounter more than a few target habitats within a single site (here, a 1 × 1 km square of the British or Irish national grid).

Another challenge for a volunteer-based monitoring scheme is the selection of a set of species that balances the demands on volunteer identification effort noted above with the need to select species indicating a wide range of ecologically informative but potentially unpredictable changes in a plant community. Random, or weighted random, selections of species can be made with respect to certain traits, coupled with filters that use information on identifiability or national frequency to eliminate very rare or critical species (Walker *et al.*, 2010b). Information on plant species associations with national vegetation classifications can also be used to restrict habitat species pools if the habitats to be monitored can be equated with these. Ultimately, where volunteer uptake is essential for the success of a scheme, we have found that the combination of the preceding steps with a group-led expert review is the most pragmatic route through species selection. Species' trait representations can be investigated post-selection to evaluate the likelihood that the final species set selected is biased in some way towards detecting, or failing to detect, certain types of environmental change.

Reporting results

An important aspect of monitoring scheme success is the timely reporting of results (Devictor *et al.*, 2010). Information obtained from monitoring schemes is increasingly reported in the form of biodiversity indicators, which are measures that are intended to illustrate trends in biodiversity over time (Vačkář *et al.*, 2012). These are often shown in an aggregated form to illustrate overall change or can be disaggregated into various sub-indices, depending on the data, to look at changes within certain species groups or across specific habitat types. Robust (i.e. unbiased and sensitive) biodiversity indicators are becoming increasingly demanded by policy makers so that assessments of human impact can be made and progress towards environmental targets can be monitored (Butchart, Walpole & Collen, 2010). The recent Convention on Biological Diversity in Aichi set five strategic goals and twenty targets with the aim of stopping biodiversity loss by 2020 (Convention on Biological Diversity, 2010). In particular, Target 12, which states that '[b]y 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained', requires the use of robust indicators to determine which species are declining, to monitor their status, and to assess whether this target has been reached. In the UK, a suite of indicators has been produced to monitor progress towards the Aichi targets and are produced annually in the 'Biodiversity Indicators in Your Pocket' report (Defra, 2014). A number of attributes are regarded as necessary for an ideal biodiversity indicator, including the need to: (1) be quantitative in that the measure is accurate with estimates of error; (2) be indicative of a wider range of species than those included within it; (3) be timely in the reporting of changes in trends; and (4) be relevant to policy (Gregory *et al.*, 2005; Jones *et al.*, 2011). Clearly, all design aspects of a scheme must be carefully considered for these attributes to be confidently defended.

IS ALL 'SURVEILLANCE' MONITORING INEFFICIENT?

Although we have discussed scheme design in the context of producing robust trends, the initial decision to commission a monitoring scheme can itself be considered as a design choice (McDonald-Madden *et al.*, 2010). The typical proposition of many atlases and monitoring schemes, namely that 'relevant, up-to-date information is essential for determining priorities and policies for conservation' (Rich & Woodruff, 1996), would appear to be uncontroversial; however, recently, a subset of the literature on biodiversity monitoring has focused on investigating the effectiveness of different types of monitoring, particularly where cost is concerned (Nichols & Williams, 2006; Wintle, Runge & Bekessy, 2010; Possingham, Fuller & Joseph, 2012b; Possingham *et al.*, 2012a). Some of this literature essentially challenges those who promote monitoring that is not specifically focused on answering pre-specified questions to justify such an approach (Yoccoz *et al.*, 2001; Nichols & Williams, 2006; Lindenmayer & Likens, 2010a); this type of monitoring has typically been referred to as 'surveillance' or 'omnibus' monitoring.

Those who are largely critical of surveillance monitoring suggest that it is too unfocused to reliably provide conclusions of scientific value to feedback into the achievement of desired goals, particularly concerning the management of habitats and species (Nichols & Williams, 2006). Question-based monitoring (QBM; sometimes also called process-based or targeted monitoring) has typically been put forward as a more desirable alternative (Table 1) (Nichols & Williams, 2006; Lindenmayer & Likens, 2010a). Critics of surveillance monitoring suggest that the stated aims of such projects could often be achieved for less effort using targeted surveys conducted by small professional teams (Nichols & Williams, 2006). Other criticisms have focused on the challenge of separating the effects of multiple drivers (Yoccoz *et al.*, 2001), pointing out that surveillance monitoring often relies on induction to reach conclusions (Table 1). Possingham *et al.* (2012a, b) have recognized a larger range of the potential benefits arising from different types of monitoring, noting that monitoring that is not specifically focused on experimental management, or on informing periodic management decision making, may still yield benefits in the areas of informing policy makers, educating or engaging the public, or detecting unanticipated ecological change. We see these last three benefits as those that most volunteer-based monitoring schemes in Britain and Europe have taken as their main motivations.

Accordingly, two of the key propositions concerning the choice between QBM and surveillance monitoring are that: (1) trend detection unlinked to action may be inefficient; and (2) surveillance monitoring could potentially be justified by the detection of 'ecological surprises' (Nichols & Williams, 2006; Lindenmayer *et al.*, 2010; Wintle *et al.*, 2010). Below, we briefly evaluate the relevance of these considerations to volunteer-based monitoring as it has proceeded in Britain and Ireland to date.

Trend detection unlinked to action may be inefficient

A monitoring scheme may identify a trend, and so a logical question to ask is: 'what next?' Unless monitoring is specifically designed to detect such a trend, then a plan to address this trend is likely to take time to formulate. Is this really a weakness of the monitoring? The link between management action and trend detection is obviously of most relevance to monitoring focusing on a limited set of processes or quantities that can conceivably be managed by a set of actions applied across the area monitored. Unfocused surveillance monitoring projects that cover large areas (e.g. those monitoring national trends), make observations at large spatial grain (e.g. atlas projects), or with broad taxonomic coverage, are unlikely to be amenable to the planning of clear actions that can be launched when certain trends are discovered, not least because of the practical problem of launching a management action across a habitat-type or landscape largely in private ownership. For example, evidence from the UK Breeding Bird Survey has revealed large declines in woodland birds, which has been linked to a decline in certain types of traditional woodland management (Fuller *et al.*, 2007). Although woodland management is not considered to be the only driver of declines in woodland birds, this knowledge could be used to promote increased woodland management as a desirable activity for avian biodiversity conservation in British woodlands; however, the launching of specific conservation activities as a result of this finding is neither direct nor straightforward.

The desire for trend detection resulting from monitoring to be always linked to management responses parallels the long-standing debate between the relative worth of observational-inductive and hypothetico-deductive studies in ecology. Nichols & Williams (2006) adduce the 'strong inference' paradigm of Platt (1964) as support for their view of monitoring, in that all monitoring should be conducted in a framework that leads to the rejection of hypotheses, allowing a gradual approach to the underlying reality. Others have pointed out that Platt's strong inference is only one possible approach to science; for example, Hilborn & Mangel (1997: 23) emphasize that 'many ecological studies are motivated by problems where such clear experimentation and "hard data" are often not possible ... or lead to other difficulties' (for a detailed treatment of these issues, see Pickett, Kolasa & Jones, 2007). Observational data

Table 1. The general features of question-based and surveillance monitoring. Adapted from Wintle *et al.* (2010) and Field *et al.* (2007). As noted by Wintle *et al.* (2010), these are the two ends of a continuum, and mixed, intermediate strategies are also possible

Project area	Question-based monitoring (QBM)	Surveillance monitoring (SM)	Additional comments
Focus	Targeted to improve management by learning about pre-specified processes May have a specific scientific purpose (e.g. monitoring optimized to discern between competing hypotheses)	Generally not based on any one particular management problem or specific scientific question	Management-targeted monitoring may be challenging for SM with multiple species and a broad scope
Design	Sampling optimal to address specific hypotheses or to estimate a state High statistical power to differentiate between specific hypotheses or to achieve precise estimates of state variables	Sampling not optimized to a particular purpose, although trend detection is often given as a rationale Potentially low power to differentiate between hypotheses or to estimate trends	Where SM is volunteer-based, power to detect a trend may be high, although we recognize that adequate volunteer engagement then becomes a key challenge
	Generally well stratified, replicated and exhibiting low bias and/or variance for the specified purpose	Often poorly stratified, not replicated, and having a biased sampling frame or one with high variance	Replication is not necessarily a problem for SM where projects have a large extent and are volunteer-based Bias in sampling frames can be addressed at the project design stage in both QBM and SM initiatives
	Variable sample sizes, depending on the question and the available resources	More often characterized by large sample sizes	–
Logical approach	Deductive reasoning	Inductive reasoning	–
	A priori hypotheses articulated	*A priori* hypotheses either not specified or vague. The hypothesis is often that certain species or habitats are likely to change in interesting ways in the future	The fundamental objective of SM is often the conservation of all species; monitoring can subsequently be focused on learning about drivers of decline (Wintle *et al.*, 2010)
Scope	Typically has a narrower geographical and/or temporal scope Fewer species, few state variables, fewer covariates	Often broad in geographical scope or long-running with no pre-defined end-point (Potentially) many species, many state variables, many covariates	–
Data collection	Generally collected by professional scientists. Expensive per data point	Generally collected by a mix of professional scientists and volunteer observers Cheap per data point	–
Potential for volunteer contribution	Likely to be restricted to the most engaged volunteers (even amateur experts in the identification of species may be deterred by rigorous sampling protocols)	Often accessible to a broad audience with varied skills, although species identification requirements may limit engagement to expert amateurs unless training is provided	–

Table 1. *Continued*

Project area	Question-based monitoring (QBM)	Surveillance monitoring (SM)	Additional comments
Public impact	Potentially less amenable as a tool for community engagement	Community engagement may be one of the primary objectives	–
Communicating key results	Straightforward where a project has clarity of purpose	Choice of result(s) typically *post hoc*, and can be challenging where resulting data are not amenable to statistical analysis	If volunteer-based SM is set up with the purpose of monitoring species of conservation interest, then the key results (trends in distribution and/or abundance) are likely to be clear

can be used to generate hypotheses, particularly if organized in a clear conceptual framework that highlights patterns; for example, see studies of biogeographical relationships (Preston & Hill, 1997) or physio-ecological trait relationships (Waite & Sack, 2010). Indeed, Proctor (2010), commenting on Waite & Sack (2010), notes that 'there are occasions when thorough exploration of an extensive block of data produces ideas and insights that no amount of hypothesis and test would have hit upon. The two approaches are complementary. The hypothesis–test approach is effective, satisfying and progressive, as long as you have a good supply of hypotheses. However, we also need systematic bodies of data, and the ideas that come from exploring them'.

Rackham (2006: 404) and Peterken & Backmeroff (1988) have made similar cases for long-term studies in woodlands. We suggest that surveillance monitoring fits into this mould (Wintle *et al.*, 2010): the patterns and data that arise from surveillance monitoring can stimulate more focused studies on the causes of ecological change. These are studies that would be unlikely to be designed in a hypothetic-deductive framework based around adaptive management, due to the resource constraints that would stop this being a reality for all but the few areas of habitat receiving regular management by conservationists. Particularly, volunteer-based surveillance monitoring enables the net to be cast wide for relatively few resources. The adaptive management/QBM paradigm is likely to be most appropriate for relatively well-resourced conservation professionals, managing populations that are amenable to management actions that can be easily deployed over a target area (e.g. the optimal harvesting of commercially valuable animals). For the rest of the landscape, we suggest that volunteer-based surveillance is likely to produce data that inform policy makers and inspire further, targeted research, or policy actions, rather than directly triggering management interventions at the level of a particular set of sites.

Surveillance monitoring could be justified by the detection of 'ecological surprises'

Wintle *et al.* (2010) have cogently argued for the wise use of surveillance monitoring, with a specific focus on the idea that one of the justifications of surveillance monitoring is the detection of unexpected patterns of change: the 'unknown unknowns', or 'surprises' *sensu* Hilborn (1987). After a short review of real and hypothetical examples of surveillance-type projects yielding new information, Wintle *et al.* (2010) attempt to derive a model that would clarify the circumstances under which investment in surveillance monitoring over QBM would be justified. This model is based within a decision-theoretic framework; it considers the relative probabilities that QBM or surveillance monitoring leads to the achievement of a management goal or to the detection of an unforeseen pattern; information on the relative rewards of achieving management goals, the costs of missing a harmful ecological novelty, and the frequency with which harmful novelties occur are also all required. The framework can in theory be used to estimate the division of funds between QBM and surveillance monitoring.

The framework of Wintle *et al.* (2010) is elegantly constructed but, in reality, the parameterization of the model would be extremely difficult. One particular obstacle in many cases would be the framing of management rewards and the costs avoided by detecting an unforeseen event: one could imagine a targeted management experiment for a particular species yielding population increases but, for most mandated surveillance monitoring, it is much harder to link the unobserved 'cost' of an unanticipated change with subsequent action that would unambiguously result in the direct avoidance of that cost. Wintle *et al.* (2010) recognize these challenges, suggesting that 'the most tractable use of the model is as a communication tool for increasing the clarity of thought and discussion about the purpose and design of the monitoring scheme'. As they emphasize, a surveillance monitoring scheme makes choices about what is important by virtue of its existence; in cases where these choices can be narrowed down to specific species or habitats, then a question-based, targeted form of monitoring may reveal itself to be more efficient (e.g. Pullin & Woodell, 1987). However, if we were able to include additional benefits of surveillance monitoring in such a decision framework, such as the increase in a public's level of investment in environmental issues (Levins, 2003; Miller, 2005), the highlighting of large-scale trends that are not easily amenable to site-level management solutions (e.g. climate change or nitrogen deposition) or the fact that volunteer-based surveillance monitoring is likely to be more heavily subsidized by the voluntary activities of interested individuals across a large area (Schmeller *et al.*, 2009), then we might find that surveillance monitoring is preferred over QBM in a larger range of scenarios.

These points present a challenge to that part of the ecological literature that deals with the implementation of and the quantification and communication of results from government-mandated, surveillance-type monitoring, without questioning whether its logical foundations are well-justified. As previously noted, Tulloch *et al.* (2013) dealt with the question of whether atlas or monitoring scheme approaches to volunteer-based monitoring are most efficient (although their metric of impact investigated was restricted to scientific citations). The more general question of whether funding agencies should support these types of volunteer-based surveillance monitoring was not specifically addressed, although it is highly likely that such activities represent excellent value for money (Schmeller *et al.*, 2009), even though such monitoring does not typically focus its activities on QBM. Possingham *et al.* (2012b) make the case for greater attempts to formalize the various benefits arising from monitoring activity that has the primary goals of informing policy makers, educating, or detecting unexpected ecological change; these authors speculate that, in schemes where these benefits increase over time, cost-effectiveness will also increase. Possingham *et al.* (2012b) conclude that a greater effort should be made to quantify these benefits, with the aim of building up a body of data that allows 'conservation returns on investments' to be routinely calculated and compared between projects, regardless of how those returns are garnered. Of course, such an activity would itself require the monitoring of a scheme's impacts (e.g. on awareness, engagement or its detection of ecological surprises) using some formalized tool or framework.

For a volunteer-based organization engaged in biological recording and natural history, mobilizing volunteers nationally on relatively straightforward projects may often be a much more likely scenario than participation in focused hypothesis- or question-driven monitoring of the type that is often framed as a direct rival to surveillance monitoring (Nichols & Williams, 2006). Although it may appear that we have set up a straw man in subjecting volunteer-based atlas projects and monitoring schemes to criticisms often directed at professional-led surveillance monitoring, these types of citizen science projects still fall into this category, and therefore the criticisms are still relevant: for example, that volunteers are wasting their time collecting information with little conservation relevance, or that the collected data are not as good as they could be for the effort expended. The recognition that other benefits should be formally taken into account (Possingham *et al.*, 2012a, b) should be welcomed as a useful addition to the debate, and one that is applicable to a broader range of monitoring activities; however, we should also be aware that formalizing the collection of further data on broader impacts may itself drain conservation resources. Lindenmayer (2012) suggests that an efficient way forward would be to find novel ways to combine data from both surveillance and question-based monitoring. The increasing popularity of hierarchical Bayesian modelling for combining data collected at different scales and with different observation processes is likely to prove useful for this purpose (Pagel *et al.*, 2014).

WHITHER VOLUNTEER-BASED SURVEILLANCE MONITORING?

A range of opinions concerning the utility of surveillance-type, long-term ecological monitoring schemes exist. Many of the strongest objections come from those that espouse more focused, hypothesis-led or adaptive-management driven monitoring, and who put forward reasonable doubts about its general cost-effectiveness. We suggest that each type of monitoring is complementary to the other, rather than being mutually exclusive: in the UK, key drivers of habitat change, nitrogen deposition for example, have been demonstrated by focused small-scale research (Stevens *et al.*, 2004), as well as by large-scale, volunteer-based recording (McClean *et al.*, 2011). The British and Irish experience suggests that flagship, volunteer-based, long-term recording and monitoring can contribute to a set of foci for national conservation (Defra, 2014), and an increased awareness by people of their local environment, as long as continual efforts are made to engage new audiences. These may then translate into new funds for conservation and a greater desire for land management that supports biodiversity (Miller, 2005; Pretty, 2012). These impacts are hard to capture, and citation-counting exercises are unlikely to do them justice.

G. C. Druce (1932), in his *The Comital Flora of the British Isles* (a list of which plants occurred in which counties of Britain and Ireland, compiled using data collected by professional and amateur botanists) was already commenting on the uses of such a list for documenting local extinctions, noting that he had 'shown ... how many plants [had] been lost to our Flora in the past', and concluding, 'I shudder to think of the tale which may be shown even in the year 2000'. Druce's prediction has largely been borne out, and further data collected, again, by a combination of professionals and expert amateurs have provided the evidence documenting impoverishments of large parts of Britain's plant diversity (Preston, 2000; Preston *et al.*, 2002c; Walker *et al.*, 2009; McClean *et al.*, 2011). Statements, then, such as those that claim we are 'just beginning to see the multiple benefits of using data from citizen science programs to monitor changes in the environment' (Tulloch *et al.*, 2013) can be seen to lack a historical perspective. The data that have allowed conclusions to be drawn about the current state of our plant biodiversity have accumulated gradually over time, and have largely been the result of curiosity-driven surveillance monitoring, combined with occasional, more structured projects (Rich & Woodruff, 1990, 1996; Braithwaite *et al.*, 2006); these have often been organized by largely volunteer-based organizations that would be highly unlikely to turn their focus completely towards conservation management. It is very likely that continued input from volunteer experts in numerous areas of surveillance monitoring will continue to produce datasets that are highly valued by governments, scientists, and the volunteers themselves (but see Ellis & Waterton, 2004), and that the development of variations on the monitoring scheme or atlas will continue to contribute as much to ecology and conservation in Britain and Ireland as they have over the past 50 years (Roy *et al.*, 2014; Pocock *et al.*, 2015).

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