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Nature, green economy and sustainable development: The outcomes of UN Rio+20 Conference on Sustainable Development

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Keywords

sustainable development, green economy, Rio+20, natural capital, ecosystem services

Introduction

The UN Conference on Sustainable Development (UNCSD / Rio+20) took place 20 – 22 June 2012, in Rio de Janeiro, Brazil. The conference was the third global high-level event on sustainable development, marking the 20th anniversary of the 1992 United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro, and the 10th anniversary of the 2002 World Summit on Sustainable Development (WSSD) in Johannesburg.

The objective of the Rio+20 conference was to secure renewed global political commitment for sustainable development, assess the progress to date and identify remaining gaps in the implementation of existing commitments (e.g. Millennium Development Goals – MDGs – agreed in 2000) (UNCSD 2012a). In addition, the conference also aimed to address new and emerging global challenges for sustainable development. In this context, the conference focused on two themes: the role of a green economy in the context of sustainable development and poverty eradication, and improving the institutional framework for sustainable development. In addition, seven thematic areas in need of priority attention were highlighted including jobs, energy, sustainable cities, food security and sustainable agriculture, water, oceans and disaster readiness.

According to the organisers, Rio+20 brought together close to 30 000 participants from across the world, making it the most attended conference in the history of the UN (UNCSD 2012a). This included representatives from over 190 UN Member States, including close to hundred Heads of State, Vice Presidents and Prime Ministers. In addition, close to 4000 representatives of media and over 10 000 representatives of NGOs and other major groups were present.

Key outcomes of the conference

The key outcome of the Rio+20 conference was the adoption of a global political agreement by the Heads of State and Government and high level representatives to renew their commitments to sustainable development and poverty eradication, and to ensure the promotion of economically, socially and environmentally sustainable future for both current and future generations (UNCSD 2012b). The agreement (called "The Future We Want" declaration) also acknowledged that since 1992 there have been areas of insufficient progress and setbacks in the integration of the three dimensions of sustainable development (i.e. economic, environmental and social sustainability) into political agendas. These setbacks have been aggravated by multiple financial, economic, food and energy crises, which have threatened the ability of all countries, in particular developing countries, to achieve sustainable development. Consequently, it was seen crucial that countries would not backtrack from their commitments made in and since 1992.

One of the most awaited elements of the Rio+20 declaration was the agreement on green economy in the context of sustainable development and poverty eradication. A green economy is commonly defined as a low carbon, resource efficient and socially inclusive economy that aims to improve human well-being and social equity while significantly reducing environmental risks and ecological scarcities (UNEP 2011). After long negotiations (e.g. prior to Rio), countries finally agreed to consider green economy as one of the important tools available for achieving sustainable development and eradicating poverty. It was, however, stated that green economy policies would need to respect each country's national sovereignty over their natural resources taking into account national circumstances, objectives, and policy priorities. In other words, no agreement was reached regarding general global rules or roadmap(s) for green economy. These rather vague and national level driven commitments were a disappointment to many participants, including the EU, who would have welcomed more concrete and rigorous global commitments related to the transition to green economy (EU press 2012).

In terms of nature, maintaining the healthy functioning of the Earth's ecosystems (e.g. removing unsustainable patterns of production and consumption that undermine biodiversity conservation) is mentioned as one of the purposes for green economy.

In addition, the direct dependency of people - especially the poor - on ecosystems and their services for livelihoods, economic, social and physical well-being, and cultural heritage is emphasised. Disappointingly, however, no specific reference is made to "greening" the existing monitoring and accounting systems for green economy, including the importance of integrating the (non-market) values of ecosystem services into national accounting frameworks. Also, no links are made to the multiple ways how working with nature (i.e. investing in so called natural capital) can proactively support the transition to green economy (ten Brink et al. 2012). Finally, there is no new commitment to removing economic incentives undermining the sustainable use of natural capital, including eliminating environmentally harmful subsidies, beyond the explicit reiteration of existing calls for reforming fossil fuel and fisheries subsidies.

In terms of institutional framework and intergovernmental arrangements for sustainable development, countries acknowledged the vital importance of an inclusive and transparent multilateral system for better addressing challenges for sustainable development and emphasised the need for an improved and more effective institutional framework (e.g. the need to promote and strengthen the effectiveness and efficiency of the UN system). As concrete outcomes, an agreement was reached to establish a universal intergovernmental high-level political forum, building on and replacing the Commission on Sustainable Development. The aim of this high-level political forum is, among other things, to provide political leadership, guidance, and recommendations for sustainable development and follow up and review progress in the implementation of sustainable development commitments.

The Rio+20 participants also reaffirmed the need to strengthen international environmental governance within the context of sustainable development. To support this objective a decision was made to strengthen the role of UN Environment Programme (UNEP) as the leading global environmental authority responsible for setting the global environmental agenda. This included, for example, agreeing to strengthen UNEP's governance structure, responsiveness and accountability to Member States and to guarantee secure, stable, adequate and increased financial resources for the programme from the regular budget (including UN budget and voluntary contributions). While a disappointment to many of those who had hope for an "upgrade" of UNEP into a fully established UN institute (World Environmental Organisation), the political agreement to reinforce UNEP's institutional standing was nevertheless seen as a step to the right direction.

In addition to the above, a framework for thematic future action and follow-up was discussed and agreed in the meeting, building on the previous commitments. The identified key focal areas for action included, for example, poverty eradication, food security, water and sanitation, energy, sustainable tourism, transport, sustainable cities, health and population, jobs and employment, risk reduction, climate change and forests and biodiversity. In this context, a specific attention was given to oceans and seas where a number of commitments were reaffirmed or made, including a commitment to intensify global efforts to meet the 2015 target to maintain or restore fish stocks to levels that can produce maximum sustainable yield.

The international commitments for conservation of biodiversity (e.g. the global Biodiversity Targets for 2020 adopted in Nagoya in 2010, so called Aichi Targets) were reiterated, emphasising both the intrinsic value of nature and its role in underpinning socio-economic development while highlighting the importance of biodiversity conservation, enhancing habitat connectivity and building ecosystem resilience. In addition, the conservation (or restoration) of biodiversity, ecosystem and related services was recognised as an integral part of action on food security and sustainable agriculture, water supply and sanitation, and sustainable development of mountain regions. Unfortunately, however, the Rio+20 outcome document falls short in highlighting important synergies between nature conservation and a number of other key areas, including the role of well-functioning ecosystems in supporting mitigation of and adaptation to climate change and reducing environmental risks. Also, there is no mention of nature's role in developing sustainable tourism and green jobs and maintaining mental health.

Finally, building on the above, an agreement was reached to complement the Millennium Development Goals (MDSs) adopted in 2000 with a set of dedicated goals for sustainable development (SDGs). While no concrete goals were established in Rio, a decision was made to establish an inclusive and transparent intergovernmental process for developing SDGs. Let by an intergovernmental committee, comprising thirty experts nominated by the five UN regional groups, the process of developing SDGs is foreseen to be concluded by 2014.

Conclusions: it is up to us to shape the future we want

While the UN and government representatives have tried their best to portray the rather timid political commitments in the best light possible, the outcomes of Rio+20 have been greeted with a wave of unveiled disappointment by NGOs and other civil society groups (e.g. Greenpeace 2012, Oxfam 2012, WWF 2012). The agreed Rio+20 declaration have been heavily criticised for the lack of concrete (new) actions and timelines. In addition, several stakeholders have raised their concern over the (seemingly) increased focus on sustainable growth instead of sustainable development in the declaration text.

The meagre global political outcomes and other concerns have led to a common consensus that the future progress on sustainable development will largely depend on actions taken by individual countries, blocs (e.g. the EU), companies and others. For example, while the EU in broad terms welcomed the Rio +20 declaration it also acknowledged that a number of its ambitions, including more concrete commitments on green economy and establishing an UN organisation for environment, were not fully achieved (EU press 2012).

Fortunately, however, the broader developments in the context of Rio+20 indicate that, regardless of the meagre global political outcome, there is a wide ranging interest in taking concrete actions towards more sustainable future. For example, hardly any companies and businesses were present in the first UN conference in 1992 whereas twenty years later they were a prominent part of the conference crowd, e.g. responsible for organising several of the over 500 side events during the conference.

In addition, a significant number of new commitments were made to complement existing global endeavours for sustainable development. For example, over fifty countries and close to ninety private companies committed to the World Bank initiative on developing natural capital accounts to support green economy, e.g. exploring the integration of (key) ecosystem services into accounting frameworks (WAVES 2012). Furthermore, more than 50 billion USD was pledged by private investors to help to implement the UN chief Ban Ki-moon's "Sustainable Energy For All" initiative (Sustainable Energy Initiative 2012). All and all, Rio+20 process resulted in close to 700 voluntary commitments for sustainable development, mobilising more than 513 billion USD worth of funding from government, business and civil society groups. These voluntary commitments cover a range of areas including energy, transport, green economy, disaster reduction, desertification, water, forests and agriculture (UNCSD 2012a).

The true key to success of Rio+20 is whether the above commitments will also be realised and whether the "leading by example" encourages others to follow suit and also develop partnerships to help address the inter-linked environmental, social and economic challenges. For example, the (already started) transition to a green economy in the context of sustainable development and poverty alleviation has not stalled at Rio, but neither has it been catalysed and accelerated sufficiently. Similarly, while the global targets for biodiversity were reaffirmed in Rio a range of concrete activities remains to be taken to ensure that these targets are met by the 2020 deadline. Therefore, what is needed is more conviction, more commitments and more implementation.

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Biodiversity conservation across scales: lessons from a science–policy dialogue

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Abstract

One of the core challenges of biodiversity conservation is to better understand the interconnectedness and interactions of scales in ecological and governance processes. These interrelationships constitute not only a complex analytical challenge but they also open up a channel for deliberative discussions and knowledge exchange between and among various societal actors which may themselves be operating at various scales, such as policy makers, land use planners, members of NGOs, and researchers. In this paper, we discuss and integrate the perspectives of various disciplines academics and stakeholders who participated in a workshop on scales of European biodiversity governance organised in Brussels in the autumn of 2010. The 23 participants represented various governmental agencies and NGOs from the European, national, and sub-national levels. The data from the focus group discussions of the workshop were analysed using qualitative content analysis. The core scale-related challenges of biodiversity policy identified by the participants were cross-level and cross-sector limitations as well as ecological, social and social-ecological complexities that potentially lead to a variety of scale-related mismatches. As ways to address these challenges the participants highlighted innovations, and an aim to develop new interdisciplinary approaches to support the processes aiming to solve current scale challenges.

Keywords

Biodiversity conservation, environmental policy, governance, scale sensitivity, scale challenge, stakeholders, academia, EU

Introduction

The year 2010 marked the deadline for the political targets to significantly reduce and halt biodiversity loss at global and the EU levels, respectively. Despite the efforts to date, assessments from global to local levels still document significant losses of diversity across spatial and temporal scales with potentially serious consequences in terms of provision of ecosystem services (GBO3 2010). Acknowledging the failure to achieve the 2010 targets, a set of new conservation targets, the Aichi targets of the Convention on Biological Diversity (CBD), have been adopted for the period 2011–2020, by the international community, including the EU (CBD 2010, EC 2010, Mace et al. 2010). In addition, preventing the degradation of wider ecosystems and their services has been incorporated in both global and the EU agendas in order to reach the set targets by 2020. Successfully meeting these ambitious targets requires critically reviewing existing and emerging biodiversity policies to improve their design and implementation based on the lessons learned.

Mismatches between the scales at which ecological processes take place and the levels at which policy decisions and management interventions are made are amongst the main shortcomings of current biodiversity policy regimes (Crowder et al. 2006, Cumming et al. 2006, Folke et al. 2007) and can be considered as one of the main reasons why the 2010 targets have not been achieved so far (Planet under pressure 2012a, b). The policies and decisions that shape human activities driving biodiversity change operate at many administrative levels, employ a range of instruments at different scales, and involve a variety of governmental and non-governmental actors (Young 2002, Young et al. 2005). These actors often have different insights in to what constitutes a scale-challenge and how to deal with it, inevitably leading to contrasting opinions. Because of this divergence of views, deliberative discussions between stakeholders provide a promising way to identify options to overcome current scale-related challenges. Accordingly, the SCALES project (Henle et al. 2010) organized a stakeholder workshop with governmental and non-governmental actors in Brussels (21/09/2010). The goal of the workshop was to encourage science-policy dialogue and to share opinions and perspectives on scale challenges and scale mismatches, between and among representatives from EU and national administrations, including ministries, environmental NGOs, and academics of various disciplines with working experience at national and EU levels. This paper provides an overview of the outcomes of the workshop, and suggests some directions towards meeting scale challenges and reducing scale mismatches.

Theory

Scale has been used in numerous ways: by referring to various sizes (small and large), to hierarchical structures composed by different levels, and to non-linear relationships taking place between and within various levels (Sayre 2008). In this paper, following Gibson et al. (2000) and Cash et al. (2006), scales are explored as simultaneously occurring dimensions (e.g., spatial and temporal) describing entities (e.g., levels of biological organisation, levels of governance systems) that have evolving interactions between each other. This approach is useful to analyse and support multi-level governance of biodiversity conservation, as it facilitates the understanding of how ecological processes and societal decisions and actions take place at, and across, many different scales. Additionally it highlights how scale-dependent these processes are, and how better decisions and improved practices could be developed (Cash et al. 2006, Gibson et al. 2000).

Different ecological processes and ecosystem functions occur at different temporal and spatial scales (Peterson et al. 1998). For example, habitat requirements for species may change with scale (Altmoos and Henle 2010) and understanding the viability of populations requires an assessment of processes at different scales (Kleyer et al. 2007). Likewise, the time lags in responses of species to fragmentation may be considerable (Henle et al. 2004). Moreover, biodiversity loss and ecosystem change are outcomes of multiple direct and indirect drivers that operate simultaneously and interactively at various scales. Some drivers that directly affect biodiversity show high scale sensitivity, i.e., they are spatially differentiated across administrative levels (Moss and Newig 2010). Characteristic example of such scale sensitivity is wetland loss, which in the EU shows a strong spatial unevenness at the national level, predominantly concentrated in central and eastern EU countries, but seems to be a much more widespread phenomenon when it is observed at lower administrative levels (at sub-national or local levels) (Figure 1). Analysing, understanding, and overcoming these ecological scale-sensitivities requires combining ecological knowledge with information, awareness and experience of actors at various governance levels thus directly bridging science and policy discourses.

Biodiversity policies do not always take into account the scale-dependence of ecological phenomena and anthropogenic activities (Henle et al. 2010). For example, the costs of conservation tend to occur at the local level, whereas benefits of biodiversity conservation, and related ecosystem services, reach far beyond municipal or private–property boundaries to regional, national, or even global levels (Perrings and Gadgil 2003, Ring and Schröter–Schlaack 2011, Santos et al. 2012). Policies and measures for their implementation are also often inadequately coordinated across geographical regions or

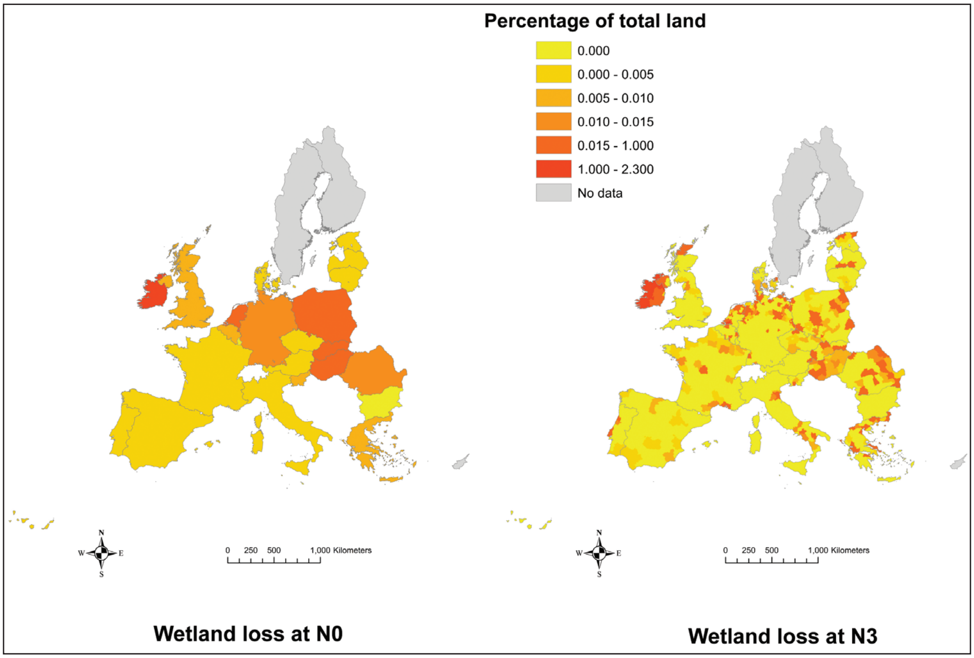

Figure 1. Changes in evenness of drivers: loss of wetlands across the EU. Abbreviation NUTS (Nomenclature of Territorial Units for Statistics) refers to the regional classification within the EU, from country level (NUTS 0) to small regions (NUTS 3). The numbers show the hectares of wetland loss as a percentage of the total land in the respective NUT.

administrative levels (Cash et al. 2006). For example, the implementation of existing key EU policy instruments for biodiversity conservation, i.e. the Birds and Habitats Directives and national laws for nature conservation, tend to focus on ensuring conservation of specific 'ecological units', (e.g. primarily protecting particular species or habitats in distinct areas without paying enough attention to wider spatial scales and the broader social-ecological systems relevant for conservation efforts (Paloniemi and Tikka 2008, Grodzińska-Jurczak and Cent 2011, Apostolopoulou et al. in press). Similarly, temporal mismatches constitute a significant scale challenge, for example time scale of biodiversity conservation does not always match with the fixed electoral cycles or the tendency of governance systems to respond to immediate, short-term economic interests.

To add to the above described complexity, coordination between different policy sectors and jurisdictional levels has often proved to be inadequate. Characteristically, biodiversity governance still has little impact on other policies influencing economic activity and land use, such as the Common Agriculture Policy (CAP), Common Fisheries Policy, transport, planning or energy policies. Policies distantly related to biodiversity conservation often have goals contradictory to safeguarding biodiversity; for instance, a governmental priority for development plans has in many occasions resulted in planning policies which hinder the enforcement of conservation measures and sustainable land use rules (Apostolopoulou and Pantis 2009). Even though some more recent policies, such as the EU Water Framework Directive (WFD), the Marine Strategy Framework Directive (MSFD) or the EU directives on Impact and Strategic Environmental Assessment, aim to integrate biodiversity conservation into other policy sectors, the actual success of such policy reforms depends on the degree of 'fit' with existing institutional structures and practices (Moss 2004). Solving these multi-level and cross-sector challenges requires, inter alia, the active participation and involvement of stakeholders from different sectors as well as from different jurisdictional and societal levels.

Material and methods

To foster an open science–policy dialogue and to explore topical and innovative ideas for better integration of scale-related issues into biodiversity policy and governance in the EU and Member States, we invited 23 stakeholders to an expert workshop in Brussels. The participants were selected to establish a diverse group of stakeholders, covering both Member States and EU level. These participants included representatives from different Directorates-General (DGs) of the European Commission, from a number of environmental NGOs operating at EU level, and from Finland, France, Germany, Greece, Poland and UK. The national level participants were from ministries, national level NGOs, and sub-national level agencies implementing biodiversity policy.

We divided the stakeholders into four small groups, each including approximately 6–7 participants from the EU level institutions and from several Member States. Each group participated in two deliberative discussions, following brief introductions to the aforementioned scale issues. The first discussion explored how effectively existing policies address scale-related issues (at EU, national and sub-national levels). The second discussion aimed to explore new policy solutions for addressing the identified scale-related challenges. The discussions were facilitated and documented by some of the authors (2 researchers were participating in each group).

The discussion topics addressed scales, and whether it was a neglected issue in current biodiversity policy and governance, what were the key reasons for and barriers to addressing scale-related issues, and what the biodiversity challenges post-2010 are with specific attention to how addressing scales could help overcome the problems identified. Each discussion lasted c. 90 minutes and all discussions were recorded and reported by taking extensive notes. The discussions were analysed following the method of qualitative content analysis (Miles and Huberman 1994): a list of viewpoints was compiled and iteratively reorganised by aggregating similar statements into broader categories until a few different categories were formed. The aim of the analysis was not to compare or count the opinions of various participants, but to find out schemes helping to solve scale challenges considered as relevant. The results of the analysis were discussed with the co-authors during the process of analysis in order to overcome possible bias.

Results and discussion

Scale challenges in biodiversity policy and governance

The dimensions of complexity

The stakeholders generally agreed that dealing with a number of different scales and their interactions simultaneously is a demanding, but important undertaking, and thus they supported the complex scale interpretation of Gibson et al. (2000) and Cash et al. (2006) even if a variety of definitions of scale concepts and challenges was reported. In particular, our analysis led to the categorization of stakeholders' perceptions into three broad dimensions of complexity: ecological, social, and social-ecological which could accordingly lead to several types of mismatches either within or across the governance scale or between ecological and governance scales.

On one hand, the participants highlighted that current policy frameworks do not possess the necessary 'scale-sensitivity' to address the inherent complexity of ecological phenomena and to take into account species and ecosystem processes that operate at different scales, and especially their relationships with fragmentation and connectivity. By underlining these aspects, the participants paid significant attention to the need to more explicitly consider ecological scale in biodiversity governance (e.g., Henle et al. 2004, Beunen and de Vries 2011).

On the other hand, the participants stressed that when considering interactions between different governance levels, it is important not to 'skip' a level but rather to take the whole spectrum of governance into consideration. In many occasions, they emphasised the complexity involved in implementing multi-level and adaptive governance approaches especially when the focus lies on the management of both social and environmental change and uncertainty across scales (Armitage et al. 2009, Leach et al. 2007). However, the participants expressed different views regarding whether scales are (or are not) necessarily always organized hierarchically or more dynamically. Some research participants argued that besides paying attention to formal administrative levels and institutions it is crucial to acknowledge that complex networks of different social groups or citizens organizations can occasionally directly link different levels, e.g. the local level may be directly linked to EU level hence by-passing the intermediate level(s). This finding concerning the role of groups and organisations that are acting between and at different levels (c.f., Swyngedouw 2004) complement the hierarchical and formal ways to acknowledge biodiversity governance. We believe that this aspect is important especially in transitional phases of governance, e.g., in periods of crises or in the framework of significant natural resource conflicts (see also Apostolopoulou and Pantis 2010).

Moreover, the participants often paid attention to the role of economic factors in the emergence of challenges in biodiversity governance. They identified the failure to link biodiversity (and its multiple values) to broader socio-economic benefits as a basis for conservation. It was argued that if biodiversity considerations are to be mainstreamed in decision making, then information about the complex roles of biodiversity and ecosystem services in supporting sustainable socio-economic systems at local, sub-national, national, and international levels should be generated and widely disseminated. This was considered as an important task, requiring a considerable amount of efforts in order to be reflected in the goals of conservation policies.

Cross-level and cross-sector limitations

The participants identified difficulties in integrating biodiversity conservation objectives set by EU, national, sub-national or local levels into the objectives and decisions at other levels. The integration of objectives between local and sub-national levels on the one hand, and the EU level on the other, was considered as especially problematic. In particular, the participants often questioned the dominant position of the EU-level actors in developing the objectives for biodiversity policy. They argued that too often local level actors were overlooked in governance processes.

In many occasions, research participants argued that the main barriers to cross-level biodiversity governance are related to structural issues and relevant 'governmental attitudes'. A recurrent statement in the discussions was that "*the EU only talks to the national level*", referring to the difficulties in incorporating EU level goals into the sub-national and local policies and vice versa. The participants also pointed out the difficulties and apparent failures in taking national characteristics into account when developing and implementing EU policy instruments. For example, while the Habitats Directive forms a legislative basis for conserving species and habitats of EU interest it does not directly provide for protecting species and habitats important at national level (e.g., nationally threatened or endemic species). It also does not take into account the specific socio-economic contexts affecting conservation in different Member States. Furthermore, the implementation of EU policies falls under the competency of the Member States or the competency of the sub-national level (e.g. the Länder level in Germany). The latter case results in a divergence between implementing institutions at sub-national levels and national levels responsible for reporting on the overall Member States' performance to the EU level, possibly leading to conflicts and confusion between actors. However, Natura 2000, the EU-wide network of conservation areas, and the main actions of National Biodiversity Action Plans, were seen in some cases as relatively successful in translating high-level aims (EU and national) into effective action at local levels. Moreover, EU policy frameworks, such as the Natura 2000 network or the Water Framework Directive, were considered by some participants as signs of a wider international reconfiguration and rescaling of power centres (including the reconfiguration of the EU's role) and decision-making processes (see also Kaika 2003).

Despite different opinions regarding the above issues, the majority of participants agreed that even when there were local-level successes, these were too infrequently 'scaled-up' efficiently to national or EU levels. Thus, the findings underline a need to pay more attention to power positions of actors acting on various governance levels and having crucial roles in supporting and/or limiting successful processes of scaling up and down (see also Chmielewski 2007, Rands et al. 2010).

The participants also asserted that the numerous problems of biodiversity conservation are related to the failure to integrate biodiversity conservation into policies that affect the drivers of biodiversity loss. They argued that, for example, agricultural policies with intended pro-conservation aims could in practice function as drivers for biodiversity loss by supporting activities harmful to biodiversity. However, some of the research participants highlighted that the reason for these perverse effects by different sectoral policies, do not primarily lie in the limited coordination across policies or administrative levels, but rather in the tensions or even contradictions between various economic interests and conservation goals (see also Rands et al. 2010). Participants often mentioned the dominant power relationships as a fundamental reason explaining the inclusion of particular interests into policy processes. These power positions, already discussed above, do challenge us to explore even more the dynamics and practices taking place within and across multi-level and multi-sectoral governance structures (Apostolopoulou et al. in press).

Possible ways towards overcoming scale challenges

In order to tackle the identified scale-related challenges of biodiversity conservation, the participants made a number of recommendations.

They called for a new approach based on a more effective combination of fixed and flexible policy objectives. In particular, they argued that there should be a balance between designing 'non-flexible' societal and ecological objectives at the EU or Member State level and providing opportunities for strengthening the adaptive capacity to deal with uncertainty and change across scales. They highlighted the need for a better balance between maintaining the core policy objectives, and providing opportunities for stakeholders to get empowered and educated and to develop innovative solutions.

The participants also recognized that responding to current policy challenges requires a context-sensitive coordination in order to combine top-down policy design and implementation with bottom-up identification of problems and solutions in biodiversity conservation. Therefore, the need for coordination across scales and sectors should not undervalue the way that historical, cultural and local conditions and customs impact on biodiversity conservation creating different needs and opportunities in different settings.

With the aim of strengthening communication, the stakeholders proposed that cross-scale communication platforms would be essential for a new 'biodiversity governance culture' with more active, equal and meaningful local participation. In particular, social learning could be encouraged by creating platforms where stakeholders from different governance levels could share concerns and solutions (c.f., Leys and Vanclay 2011). This generates a need for various social networks to work together in an integrative fashion (Olsson et al. 2007). In this context, establishing thematic networks (e.g., for combining implementation, monitoring and appraisal of relevant EU Directives and instruments) was proposed by research participants as a potential way to integrate existing activities. Also, these platforms could be crucial for developing best practice guidance and bridge the gap between the EU, national and local levels on land-use issues related to biodiversity conservation.

Finally, in order to improve cross-sector communication, the participants called for the development of new interdisciplinary approaches (Farrell et al. 2012). For example, they encouraged complementing ecological expertise with geographical and social expertise in land-use planning.

The science-policy discussion in the workshop proved to be a promising forum to present, negotiate and evaluate the research problems and findings between scientists, NGO representatives, policymakers and environmental authorities. The discussions between these actors and their results as presented in this paper illustrate a possible way of opening-up scientific discourses towards 'extended peer communities' (Funtowicz and Ravetz 1993). Opening-up scientific discourses seems to be especially relevant in the cases like biodiversity governance in which the novel scientific knowledge plays a remarkable role, but cannot solve the identified threats and problems as such, without being interpreted, evaluated and implemented in combination with the context-specific knowledge of concrete practices.

Conclusions

We analysed the perspectives of policy makers, practitioners, and researchers with the aim of understanding the variety of views on current and emerging scale-related challenges of biodiversity conservation, as well as exploring opportunities for solving them. The participants of the workshop agreed that addressing the interconnectedness and interactions of scales in different ecological and governance processes is essential for achieving the goal to reduce biodiversity loss.

Our main finding is that scale-related problems, and potential solutions, are all about increasing our understanding of complexity and implementing this new knowledge. Dealing with a number of different scales and scale-mismatches emerging in biodiversity and its governance is unquestionably challenging; it requires an analytical and political framework that enables the simultaneous assessment of drivers, pressures and impacts as well as policy processes and practices at various scales and levels. Additionally, tackling scale challenges requires concrete steps towards the integration of biodiversity policies across governance levels and policy sectors and integrative governance institutions and networks. In the workshop, cross-scale communication platforms were considered as a promising forum to support communication and social learning. However, new, context-specific ideas are still needed to build dynamic governing structures and flexible policy processes to encourage more legitimate, fair, integrative and innovative biodiversity conservation practices.

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RESEARCH ARTICLE

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Global Change Projections for Taiwan Island Birds: Linking Current and Future Distributions

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Abstract

The earth is warming rapidly. Species around the world must adapt to the increasing heat and to the rapid rate of temperature change. Decision makers and managers must aid species to adapt and to keep up with the changes if they are not able to do so on their own. Special attention needs to be paid to small islands because they are at high risk for the loss of unique and threatened systems and species, and face habitat loss as a consequence of climate-induced rises in sea level. In this study, we examined 17 endemic avian species on the island of Taiwan. Bird observations from 1993 to 2004 were compared to modeled distributions for 2020, 2050, 2080 and 2100. We used 5 general circulation models (CCCMA, CCSR, EHAM4, GFDL, and HADCM3) for the Intergovernmental Panel for Climate Change A2 and B2 scenarios. Results show that the distributions of 15 out of 17 species are predicted to shift up in elevation with warming. As the lower distributional limits contract to higher elevation, the upper edge of their current distributions cannot shift up in elevation because they were already near or at the tops of the mountains. Consequently, their distributions are predicted to shrink over time. The median elevation of each of these species' distributions is higher than the median elevation of all available habitats on Taiwan. In addition, we find that a few common species are predicted to become rare species under climate change. Two of the 17 species examined are not near the tops of the mountains and are the only species that have median elevations of their distributions lower than the median of all available habitats on Taiwan. These 2 species are predicted to expand the upper-elevation distribution limit but not to contract the lower-elevational limit, which results in a widening of their distributions. Hence, we suggest that the median elevation of a species' current distribution plays a key rule and can be further used as an index of the response birds most likely will exhibit as the temperature increases.

Keywords
Climate change, projection distribution, island species, endemism, geographic patterns

Introduction

The earth's ambient temperature is increasing at an alarming rate and magnitude. These increases are producing changes in natural systems (Schneider and Root 2002, Walther et al. 2002, Hampe and Petit 2003, Parmesan and Yohe 2003, Root et al. 2003). Root and coauthors (2005) found that anthropogenic forcing factors are the main causes of the discernible influences on biological systems, and an expansion that included physical and more biological systems is reported by the Intergovernmental Panel on Climate Change (IPCC: Parry et al. 2007) and by an update to the IPCC (Smith et al. 2009). Observations over three to five decades show that species are undergoing several different types of adaptive changes related to climate change. Distributions are shifting poleward and to higher elevations, phenological events are occurring earlier in spring, and behavior, morphology and genetics are changing (Parmesan et al. 1999, Pounds et al. 1999, Parmesan and Yohe 2003, Root et al. 2003, Butler et al. 2007, Wiens et al. 2009, Bradshaw and Holzapfel 2010, Emerson et al. 2010).

The risk of extinctions worldwide will likely be exacerbated by climate change that occurs too rapidly to allow species to adapt (Thomas et al. 2004). Island species are at particularly high risk because these species have small ranges and shifting poleward may not be an option (IPCC: Watson et al. 1996). They also face habitat loss produced by climate-induced rises in sea level (Smith et al. 2009). Predicting possible changes in the ranges of species caused by future warming of the earth can offer advance warning. Such predictions can also help policymakers and conservation managers draft viable mitigation and adaptation strategies. This knowledge could help lessen future impacts on natural systems (Wiens et al. 2009). The developments of programs or models that provide accurate species-distribution predictions have made gradual progress. Projections from these models increase our understanding of possible ecological consequences under various emission frameworks that range from non-mitigation scenarios to idealized long-term scenarios (Wiens et al. 2009).

An appropriate scale and list of environmental factors are both key factors in the prediction of species distributions. Investigating scaling questions is worthwhile on both islands and continents. The choice of a spatial scale to be used in species-distribution models plays an important role in model performance and follow-up applications. However, the available environmental data and species-occurrence records often limit the choice of scale, especially with respect to finer scales (Seo et al. 2009). By definition, averaging environmental information to determine values at a larger grid-cell size results in loss of information within the averaged area. Such loss of information often causes overestimates of potentially suitable areas and may fail to capture many habitat features, including, for example, information about microhabitats that are important to species in an area (Root and Schneider 1993, Ko et al. 2009, Wiens and Bachelet 2010). Other uncertainties, such as mismatches in an overall geographic range of a species in a specific area, may occur when the grid-cell size used to predict species distributions is large (Wiens et al. 2009). Using the finest grid-cell size possible for modeling increases the probability of capturing the factors, which determine species' realized niches (Wiens et al. 2009). Ko et al. (2009) used a one-square-kilometer grid system to compare four species distribution models in order to predict the current distributions of 17 endemic bird species on the island of Taiwan. These four models were logistic regression (LR), multiple discriminant analysis (MDA), a genetic algorithm for rule-set prediction (GARP), and an artificial neural network (ANN). That study demonstrated good predictive results at this fine spatial scale. Consequently, we adopted a one-square-kilometer grid system for all models in this study.

Selecting appropriate environmental factors for investigating species distribution models is critical when investigating the suitability of models that can be used to aid in conservation efforts for different species. Numerous studies have assessed many environmental factors. These studies have calculated and examined the individual contributions of different environmental factors and their effects on the distribution and abundance of different taxa (Moles et al. 2003, Jafari et al. 2004, Root et al. 2005). These studies provide strong evidence that current values applied to general circulation models (GCMs) can be used to predict species distribution changes.

To provide a comprehensive perspective on the effects of future warming, studies often use predicted changes in temperature and precipitation to simulate the effects of future warming. Various models of future temperature and precipitation have been created under different assumptions about the trajectory of global emissions. The variability in projected precipitation among the different models is quite a bit larger than that for projected temperature. On the island of Taiwan, many factors, including occurrences of typhoons and ocean cycling, cause large variations in precipitation. These variations coupled with the model variations make it difficult to forecast general or specific patterns in the magnitude of future precipitation. Moreover, the temperature showed higher contributions than the precipitation on distributions of the Taiwanese endemic bird species (Ko et al. 2009). For these reasons, projected temperature data from 127 stations (from the Central Weather Bureau of Taiwan) were downscaled and interpolated by multiple regression to the one-square-kilometer grid spatial resolution and precipitation data were not used in this study.

Studies linking the current and future distributions of species are expected to yield useful suggestions about possible indicators for use as reference points for conservation efforts in response to climate change. Current literature on the projection of species distributions, however, involves active debates about questions including uncertainties in species' dispersal abilities, migration rates, and area sizes of fundamental and realized niches occupied by species (Davis et al. 1998, Huntley et al. 2004, Thomas et al. 2004, Guisan and Thuiller 2005). In this study, we assumed that temperature was the only variable, which means we assumed, for example, that dispersal abilities were not in question and needed habitats were always assumed to be available anywhere on the island. This means we found the largest possible projections of the species distributions and the reality will most likely be smaller than those projected.

The island of Taiwan serves as a representative island for island studies generally. Taiwan, located at the boundary between the tropics and the subtropics, offers a range of topography and biodiversity. Global warming obviously impacts the island: from 1900 to 2010, island temperatures increased by approximately 1.6°C, an increase of 0.13–0.15°C per decade (Fig. 1). Current and projected increasing damage and irreversible loss of unique and threatened natural systems have attracted attention because Taiwan is a prioritized conservation hotspot in Asia (Myers et al. 2000, Lei et al. 2003, Lei et al. 2007). Many different types of data are available for numerous species on Taiwan. For example, data on more than 150 bird species have been collected at different locations since 1993. These data allow investigations to be carried out for different taxa at various elevations. Accordingly, the island serves as an example of processes occurring on small islands with significant topography, in general.

We assumed that species distributions will shift in elevation in concert with warming temperature. Depending on the physiological ecology of the species in question and compiling studies projecting species ranges (Walther et al. 2002, Sekercioglu

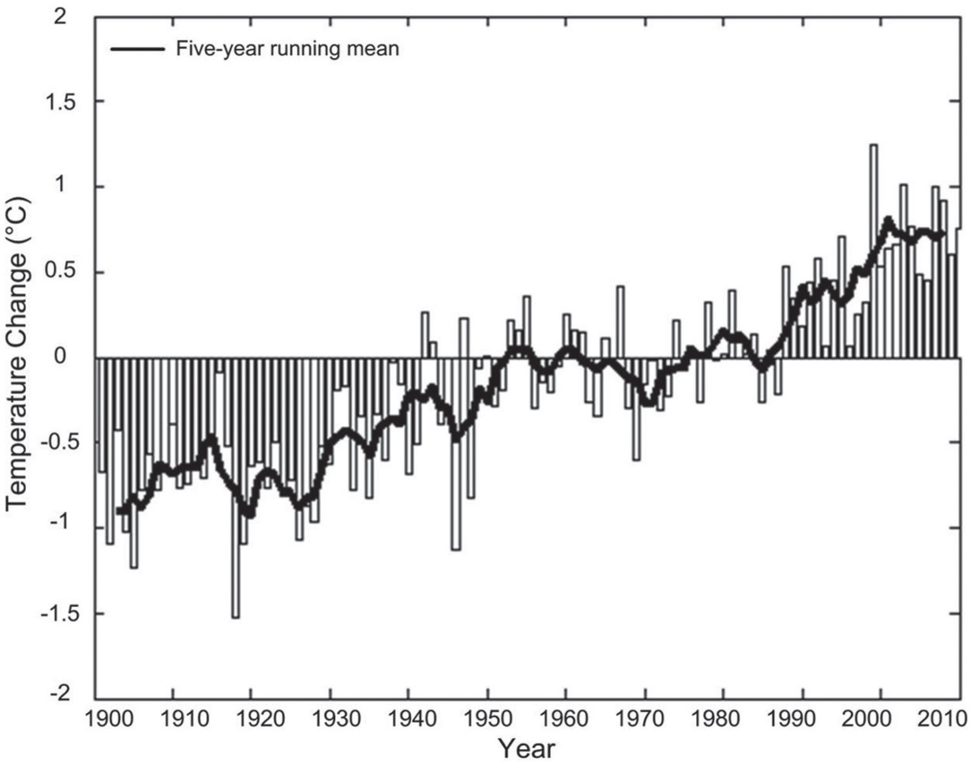

Figure 1. The increase in the observed annual mean temperature on Taiwan from 1900–2010. Temperature change in each year was calculated by comparison of the annual mean temperatures averaged from 1990 to 1999.

et al. 2008, Bradshaw and Holzapfel 2010, La Sorte and Jetz 2010a, 2010b), we made a simple assumption that species unable to tolerate a warming climate would shift up into higher elevations and perhaps contract the original ranges in lower elevations. We examined the relationship between current distributions of birds over the elevational gradient and distributions predicted as a result of future warming. The purpose of this study is to estimate how climate change affects bird species on a small island based on current associations over 12 years between species occurrences and temperature, and to identify inherent or external factors that might influence their future distributions, which can further be used as a conservation index. We focused on 17 Taiwanese endemic bird species and 10 projected temperatures (see below) to estimate possible future changes in the patterns of species distributions and to assess the relationships among these factors.

Materials and methods

Species data

We obtained occurrence data for the 17 Taiwanese endemic species from a 12-year inventory of avifauna from 1993 to 2004 (Hsu et al. 2004, Koh et al. 2006). Sampling sites were selected to represent the habitat characteristics of a particular elevation, forest type, and ecoregion. Each site, 150 m and 200m apart individually within a 1.5–3-km-long transect, was sampled once a year during the breeding season or seasonally for 2–3 years during the period 1993–2004. The data were not collected from all of the sampled sites in each census year owing to limited sampling resources. An individual site was surveyed 10–12 separate times, on average, in those 2–3 sampled years. The precise locations of the sampled sites were recorded using a Global Positioning System (GPS). These location data were transformed to a one-square-kilometer grid system. We categorized the grids as species present, absent or nonsurvey, and we ignored the abundance of individual species within a grid. If one individual was seen in a grid box, it was categorized as present. A total of 4,082 grids were surveyed, which was approximately 11% of the area of Taiwan. Due to our census restrictions, there are undoubtedly more locations where a species was present but we did not observe it, and hence a grid box could have been labeled as species absent when the species was actually present. In order to avoid those uncertain species-absent records to affect follow-up species predictive distributions, we used species-presence records only in the models of this study.

Current and future temperature data

We obtained monthly mean temperature data from the Central Weather Bureau of Taiwan form 1990 to 1999 at 25 long-term climate-monitoring stations and 102 locations where there were only temperature recording equipment and auto-rain gauges of the Central Weather Bureau of Taiwan. We averaged temperature in those 10 years in each grid cell and used these as a surface showing of current annual mean temperature. High-resolution gridded climate data are required for spatial analysis of a small region with complex terrain and diverse climate states. The data-assimilation method, a technique using statistical analysis and interpolation to integrate irregularly distributed observation into regular model grids (Wang et al. 2000), used multiple variants regression incorporating observations, coarse-grid reanalysis data, and physiographic features to generate one-km high-resolution grid temperature data (Lin and Lin in review).

To make sure the temperature data used in this study were the most suitable and representative, we compared temperature data from a widely used global climate dataset WorldClim (http://www.worldclim.org/) developed by Hijmans et al. (2005) to a temperature data set developed specifically for Taiwan by Lin et al. (2010). The current temperature differed markedly between the two datasets, especially in the 19–21°C and 22–24°C ranges. The WorldClim data contained more areas within the 19–21°C range, whereas the temperature data that we used contained more areas within the 22–24°C range. The ranges of average annual temperature simultaneously showed a significantly difference ($p<0.05$) between the WorldClim data and Lin et al.'s data, primarily caused by the two sets of data covering different time periods: 1950–2000 for WorldClim and 1990–1999 for the Lin et al. data. Comparisons to temperatures in Taiwan from Wu et al. (2008) since 1970, which covered a whole period of the avain censused years in this study, showed that the temperature of the whole Taiwan generated by the 1990–1999 period was more similar than that of the 1950–2000 periods. Thus, we adopted the projected temperature originated from the 1990–1999 temperature data by Lin et al. (2010) in this study instead of using the WorldClim dataset.

We obtained projected temperature trends, which had been specifically and locally downscaled to the 1×1km fine resolution in Taiwan by the same regression-based statistical downscaling approach (Lin et al. 2010), from five GCMs for the A2 and B2 emission scenarios from the IPCC Fourth Assessment Report: (i) CCSR: Centre for Climate System Research, (ii) CCCMA: Canadian Center for Climate Modeling and Analysis, (iii) ECHAM4: European Center for Medium Range Weather Forecasting, (iv) GFDL: Geophysical Fluid Dynamics Laboratory, and (v) HadCM3: United Kingdom Meteorology Office.

The A2 and B2 scenarios were treated as antithetical frameworks. Taken together, they provided individual alternative scenarios for ways in which the future might unfold, according to the IPCC Special Report on Emissions Scenarios in 2007. The two scenarios were derived from different assumptions about the release of greenhouse gases and aerosols into the atmosphere. They are used to explore future demographic, social, economic, technological and environmental developments on a global scale (Nakicenovic et al. 2000). The A2 scenario represents a heterogeneous world with a continuously increasing global population and regional economic growth. These levels are higher than those assumed by the B2 scenario.

The original data from the GCM scenario-run outputs, adjusted for 1990s means, were first downscaled to local weather stations. They were then further adjusted by linking the normalized probability distribution functions of deviations of the monthly mean climate parameters. Finally, a linear interpolation method was used to generate interpolated climate-change surfaces. These surfaces gave values that, added to observed base temperatures, yielded the projected temperatures. Five projected-temperature models (*i.e.* CCSR, CCCMA, ECHAM4, GFDL, and HadCM3) for each of the A2 and B2 scenarios were used to predict the study species' future distributions.

We used four temporal snapshots of the future changes in species distributions: 2020, 2050, 2080, and 2100. These years represented separate short-term, mid-term, and long-term climate-change influences on species distributions. To avoid chaotic weather fluctuations in the projected temperatures in one year, we used the average temperature from the prior 10 years for each year (*e.g.*, the temperature projected for 2020 was calculated from 2010–2019).

Model use and statistical analysis

A maximum-entropy approach (*i.e.*, Maxent) (Phillips et al. 2006), which individually analyzes the weights of environmental factors and calculates a continuous probability value for each species' distribution and has been estimated its high predictive performance among the Taiwanese species taxa (Lee et al. 2010, Ko et al. 2011), was used to project the future potential distributions. With five GCMs, two scenarios, and four snapshot years, each species distribution was predicted by a total of 40 models. We ran each model for 1000 iterations and then chose the values of the 10^{th} percentile of presence as a threshold to derive categorical values of species presence and absence from the calculated probability values. Projected-distributional probabilities of each species for each year in all models were switched to presence-absence values, which were used only in follow-up analyses. Projections of species distributions were first separated according to the two scenarios. Projected species-present grids of each species for each year from the five models in a scenario were individually normalized by calculating numbers of the grids where a species occurred compared to the total number of grids over all of Taiwan's. These proportions from all five models were then averaged by species for each of the snapshot years. Hence, each species was assigned an average proportion value for its distribution area in each year. The slope over the four snapshot years was then determined for both A2 and B2 scenarios for each species. We compared the differences between A2 and B2 scenarios with a paired t-test using SYSTAT 12.

Species' current distributions given in Ko et al. (2009, 2010) and topographical data of the whole Taiwan (Ko et al. 2010) were used to analyze species geographical changes from current and future warming temperature. The 17 endemic species generally favored habitats with high vegetation cover, at almost full forest cover and median to high NDVI, but occupied heterogeneous elevation and climatic conditions when comparing their distribution species by species in depth. In grid boxes with values greater than the 10^{th} percentile a given species was considered as species presence areas. The distribution was plotted for each of the snapshot years for each species.

Results

The projected future temperatures in Taiwan generated by the five models for the A2 and B2 scenarios showed increasing trends (Fig. 2a, 2b). The projected temperature by the five models for each temporal snapshot were different between A2 and B2 scenarios ($p>0.05$ in 2020 and 2050 and $p<0.05$ in 2080 and 2100) (Fig. 2c). In 2020 and 2050, projected B2 values were higher than those of A2, whereas the values in 2080 and 2100 showed A2 averages higher than B2. The averaged values of the projected increase in temperature ranged from 0.2°C to 3.1°C for the A2 scenario from 2020 to 2100, respectively, and the corresponding range for B2 increased from 0.4°C to 2.2°C, respectively.

Fifteen of the Taiwanese endemic bird species were projected to decrease in distribution by 2100, whereas two species, the Styan's Bulbul (*Pycnonotus taivanus*) and the Taiwan Hwamei (*Garrulax taewanus*), were predicted to increase (Table 1). A comparison of the actual observed elevation ranges and medians showed that the two "increasing-distribution" species occupied ranges whose median elevation was below the overall median elevation of Taiwan (Table 1).

Estimates of the change (*i.e.*, slope) in the percentage of the distribution areas over the four snapshot years and estimates of the lowest and highest observed elevation of individual species showed no significant difference among the seventeen species ($p>0.1$) (Table 1). The species-occurrence elevations of the 15 "decreasing-distribution" species were divided into current low- to mid-elevation and mid- to high-elevation species. The division between these two elevations was chosen to be at 1600 m based on the species' currently occupied elevation (Table 1). The current low-to-mid-elevation species (7 species) exhibited a significant decrease between 2020 and 2100 under both A2 and B2 scenarios ($p<0.01$) while no significant decrease in the current mid- to-high-elevation species (8 species) (Table 2).

The geographical changes in the species' future distributions in both the A2 and B2 scenarios indicated that the "decreasing-distribution" species would shift towards high- elevation areas and contract along warming boundaries. The result of this shift would be an overall decrease in distribution (Fig. 3). For example, the Taiwan Yuhina showed a monotonic contraction of its distributions from 2020 to 2050, to 2080, and to 2100 (Fig. 3). The general plots of all 15 "decreasing-distribution" species showed that the species would have greater distributional changes in Western Taiwan than Eastern Taiwan, owing to the relative steepness of the topography on the eastern and western sides of the island. The "increasing-distribution" species would not change their distributions by shifting their lower distributional boundary to a higher elevation. Rather, the projections show they would maintain their lower boundary and shift their upper boundary to a higher elevation. The Taiwan Hwamei (Fig. 3) is an example of this pattern of distributional shift.

Changing trends in species' projected occupancy of maximum and minimum elevations revealed differing shift rates that could further explain why a species could increase or decrease (Fig. 4). Under both the A2 and B2 scenarios, the "decreasing-

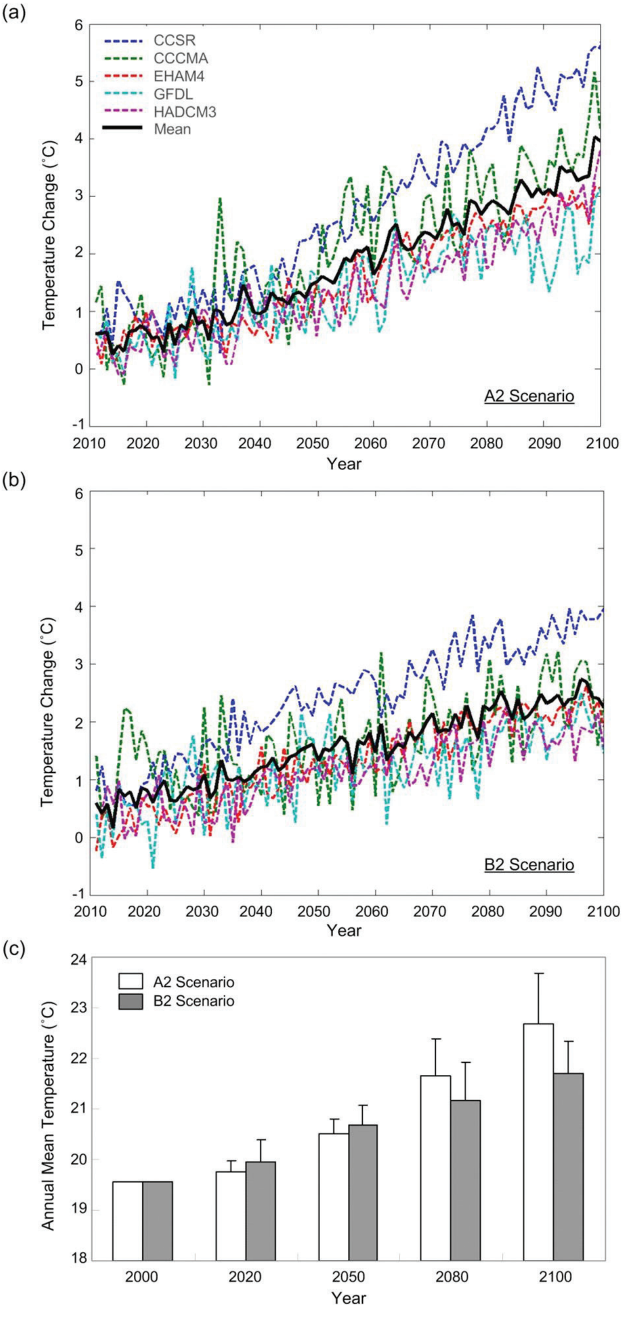

Figure 2. Changing temperature patterns in Taiwan. **a** projected temperature for 2010–2100 from five general circulation models (CCCMA, CCSR, EHAM4, GFDL, and HADCM3) under the A2 emission scenario **b** projected temperature for 2010–2100 from five models under the B2 emission scenario, and **c** average projected temperature from four temporal snapshots, 2020, 2050, 2080, and 2100. The temperature in 2000 is presented for the current climate. The temperature change is from the averaged annual mean temperature from 1990 to 1999.

Table 1. Seventeen Taiwanese endemic bird species that show changes in the % of Taiwan grip boxes where the species was present (*i.e.*, positive or negative slope) during 2020–2100 under the A2 and B2 projected scenarios and grid features of the elevation of their distributions.

English Name	Scientific Name	Increasing/ Decreasing Distribution (Slope)		Species' Currently Occupied Elevation (m)			
		A2 scenario	B2 scenario	Range	Difference	Median	IQR*
Taiwan				0-3707	3707	381	1194
Decreasing-Distribution Species							
Taiwan Barbet	*Megalaima nuchalis*	-11.0	-7.6	2–2956	2954	415	629
Formosan Whistling-Thrush	*Myophonus insularis*	-5.6	-3.4	7–2764	2757	720	890
Taiwan Partridge	*Arborophila crudigularis*	-5.5	-3.3	18–2630	2612	1125	933
Formosan Magpie	*Urocissa caerulea*	-4.3	-2.8	7–1487	1480	406	439
White-eared Sibia	*Heterophasia auricularis*	-4.3	-2.5	7–3358	3351	1425	969
Swinhoe's Pheasant	*Lophura swinhoii*	-4.1	-2.4	100–2457	2357	1389	760
Taiwan Yuhina	*Yuhina brunneiceps*	-4.1	-2.4	7–3358	3351	1587	893
Steere's Liocichla	*Liocichla steerii*	-3.3	-1.9	98–3155	3057	1639	799
Taiwan Bush-Warbler	*Bradypterus alishanensis*	-3.2	-1.9	147–3422	3275	2179	969
Collared Bush-Robin	*Tarsiger johnstoniae*	-2.9	-1.8	100–3707	3607	2284	699
Yellow Tit	*Macholophus holsti*	-2.9	-1.6	7–2815	2808	1622	615
White-whiskered Laughingthrush	*Garrulax morrisonianus*	-2.8	-1.7	100–3707	3607	2338	706
Taiwan Barwing	*Actinodura morrisoniana*	-2.8	-1.6	7–3015	3008	2102	597
Mikado Pheasant	*Syrmaticus mikado*	-2.8	-1.6	100–2979	2879	2121	623
Flamecrest	*Regulus goodfellowi*	-2.6	-1.7	378–3707	3329	2573	654
Increasing-Distribution Species							
Styan's Bulbul	*Pycnonotus taivanus*	3.9	1.6	3–2321	2318	169	259
Taiwan Hwamei	*Garrulax taewanus*	3.2	1.2	2–2735	2733	276	420

*IQR = interquartile range

distribution" species' shift in minimum elevations from 2020 to 2100 would occur more rapidly than the corresponding change in maximum elevations. Together, these changes would cause species' distributions to shrink (Fig. 4a). However, the "increasing-distribution" species would have an unchanging minimum elevations but higher maximum elevations than in a previous temporal snapshot. These changes would broaden their distribution areas (Fig. 4b).

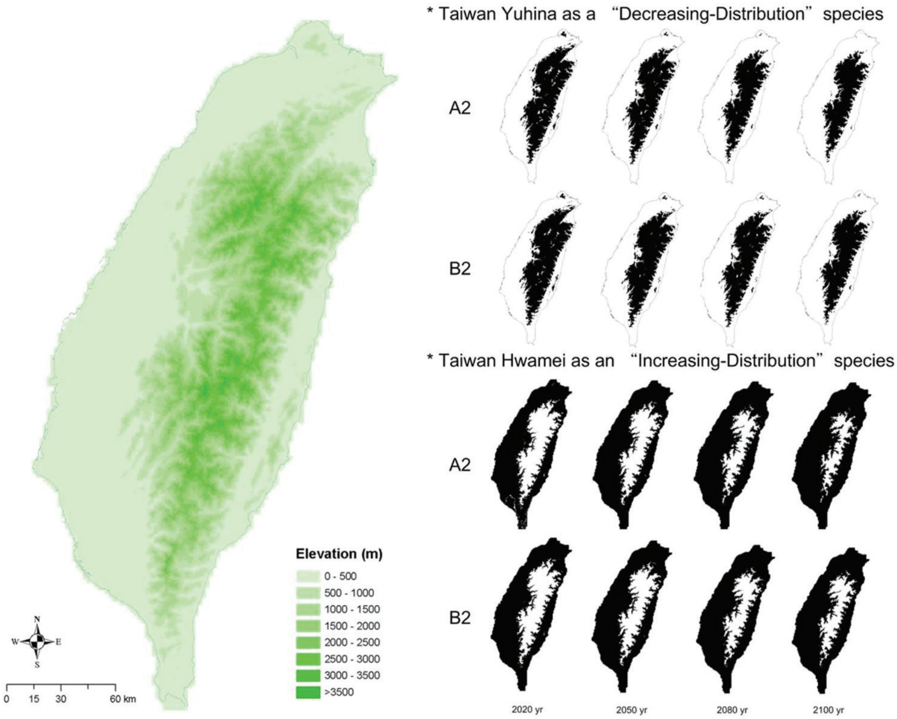

Figure 3. Elevation maps of Taiwan and geographic changes with warming temperature in the projected distributions of Taiwanese endemic bird species in 2020, 2050, 2080, and 2100. The Taiwan Yuhina (*Yuhina brunneiceps*), an example of a "decreasing-distribution" species (the right-hand upper eight maps), would decrease in distribution as it shifts to higher elevations. The Taiwan Hwamei (*Garrulax taewanus*), an example of an "increasing-distribution" species (the right-hand lower eight maps) would increase its distribution by moving to higher elevations. The black areas are species' projected-present distributions, whereas the white areas are species' projected-absent distributions.

Discussion

Anthropogenic climate change is causing an increase in temperatures in Taiwan. Our results indicated that the 17 endemic bird species could have changes in their distributional range as a result of continued warming. All 17 species show changes in the upper elevational distributional limits shifting upward. The species with their medians higher than that of all of Taiwan would contract their lower-elevation boundaries with increases in temperature, while species with lower medians than that of all of Taiwan would extend their higher-elevation boundaries but not contract their lower boundaries. Because this work was performed on an island, we were able to examine the changes in the percentage of the distributional areas for each species.

Table 2. Differences in the percentage of future projected species present in areas of Taiwan between 2020 and 2100 for 15 endemic bird species that showed an expected decrease in distribution as result of climate change. The species' distributional altitudes were based on occurrence records using the 1600 m median elevation as a threshold for determining current low-mid- (LM) and mid-high-altitude (MH) species. The current conservation status of a species was characterized as common (C), uncommon (U) or rare (R), as defined by Ko et al. (2010), depending on the number of grids in which a species was seen.

English Name	Distributional altitude	Current Status	% of Distribution Areas Decrease between 2020 and 2100	
			A2 scenario	B2 scenario
Taiwan Barbet	LM	C	32.1	23.6
Taiwan Partridge	LM	C	16.0	10.4
Formosan Whistling-Thrush	LM	C	16.3	10.8
White-eared Sibia	LM	C	12.2	7.6
Taiwan Yuhina	LM	C	11.6	7.3
Formosan Magpie	LM	U	12.8	8.9
Swinhoe's Pheasant	LM	R	12.0	7.8
Steere's Liocichla	MH	C	9.4	5.7
Collared Bush-Robin	MH	C	8.5	5.4
White-whiskered Laughingthrush	MH	C	8.1	5.2
Yellow Tit	MH	U	8.4	5.0
Taiwan Barwing	MH	U	7.9	4.8
Taiwan Bush-Warbler	MH	U	9.2	5.8
Flamecrest	MH	U	7.7	5.0
Mikado Pheasant	MH	R	8.0	4.9

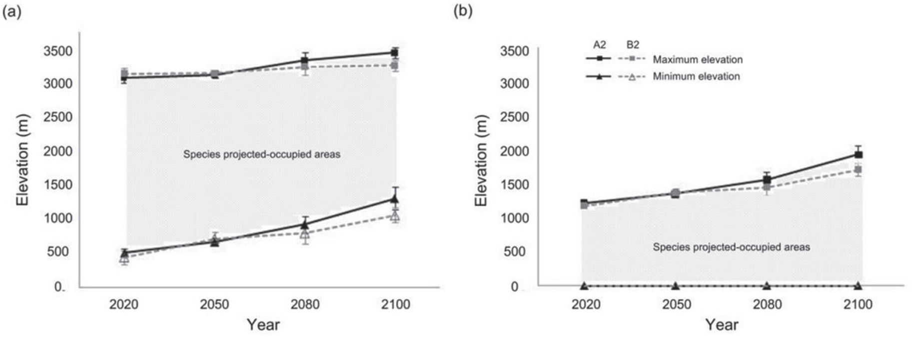

Figure 4. Changing trends in species' projected-occupied maximum and minimum elevations. A species **a** with faster shift rates at minimum elevations than at maximum elevations would become a "decreasing-distribution" species (example: Taiwan Yuhina), and **b** those with a stable minimum elevation and an increasing maximum elevation would become an "increasing-distribution" species (example: Taiwan Hwamei) in response to climate change.

As the temperature warms, the assemblages of species present in a given place are likely to change over time (Butler et al. 2007). This change in distribution raises several issues such as if these 17 endemic species will maintain competition within and among species and, more importantly, whether food resources (*i.e.*, plant and insect taxa) will change in distribution at the same time or if the species change their diets. The interactions that lead to shifts in species distributions are complicated, and the differential species' responses to temperature at multiple temporal and spatial scales are important areas for further investigation.

Potential expansion of species distributions

Using current environmental correlates of a species' distribution to project its future occurrence by using five different models assumes that the species would exhibit the same behavior regardless of the type of habitat into which their distribution would expand. This assumption could certainly hold for some but not all species. This caveat must be considered in the case of the two species we examined the projections that showed expansion of their distributions. Indeed, the projected expansion may not be possible because of the type of habitat available. Additionally, land-use change could significantly imperil the ability of a species to expand its range (Lubowski et al. 2006, Feeley and Silman 2010). The models that we used in this study did not take into account the possibility of habitat change with elevation or that of land-use change.

The projected expansion could indeed be overly optimistic about places in which the species could inhabit. Various ecological and geographical barriers across space are other key points, in addition to climate, which affect species dispersal and distribution and habitat connectivity. For instance, the Styan's Bulbul is currently restricted in distribution to a narrow area of eastern Taiwan. According to its projected geographical pattern, this species could expand to western Taiwan and become an "island-widespread" species instead of a "regionally-widespread" species. This expansion would require that the species increase its distribution across or around the Central Ridge Mountains of Taiwan in response to climate change. According to mitochondrial DNA (mtDNA) research on phylogeography in Taiwan show several species, such as *Japalura swinhonis*, *Rana limnocharis*, *Gekko hokouensis*, *Mus musculus*, *Rhacophorus taipeianus*, and *Rhacophorus moltrechti*, are divided into multiple small distributional populations by the Central Ridge Mountains (Hall and Holloway 1998, Cox and Moore 1999). This natural geographical barrier has obstructed gene flow among the populations, which further lead to species differentiation. Thus, the Central Ridge of Maintains plays a key role in keeping species from expanding their distributional ranges. This could be the case in the other two species examined here.

In addition, based on the projected distributions, the Styan's Bulbul and the Taiwan Hwamei will likely occupy their original low-elevation habitats in addition to expanding to higher-elevation areas. At some point, however, the temperature could become so warm that the low-elevation heat could lead this species to abandon its original lowland habitats and move up in elevation.

Potential shifts in species distributions

The above-mentioned caveats notwithstanding, our results suggest possible elevational changes owing to increasing temperatures. Based on the observed records, the 17 species had ranges spanning 1480m in altitude for the species with the narrowest elevational range. The widest range is 3607m (Table 1). The species at these two extremes showed the same type of distributional change. The distribution of the species with the narrowest range (*i.e.* Formosan Magpie, *Urocissa caerulea*) is projected to decrease in size by 12.8% and 8.9% in the A2 and B2 scenarios respectively. The distributions of three other species had large decreases in size even though they had wider elevational ranges (Table 2). This result does not entirely agree with recent findings by Sekercioglu et al. (2008) and Harris and Pimm (2008), who mentioned that species with wider elevational ranges were less likely to be threatened by global climate change. In comparisons with La Sorte and Jetz's findings (2010b), our results were much similar to simulated species being treated under a constrained vertical dispersal scenario than global montane birds in their study, but however, also not exactly the same. Their neutral patterns showed the simulated species with small vertical range extents (*i.e.* elevational range) contained median range losses of 100 per cent while of those species with large vertical range extents remained more of range sizes. The Taiwanese endemic bird species with small elevational range showed range decreasing, as La Sorte and Jetz's findings, but some species may not be able to remain more range sizes, even through the species with large elevational range. Importantly, the Taiwanese endemic bird species do not fit their definition for montane bird species which may lead to the above differences. The specificity of distributional changes of island species when facing climate change is valuable to future exploration. The location and structure of mountain systems, especially an island like Taiwan, come out as a strong drive of extinction risk, which may beyond our expectations on species range shifts in a warming world.

The median elevation occupied by species, relative to the value of median elevation for Taiwan as a whole, exhibits a pattern of changes in the projected future distributions. The species with their median elevation lower than the overall medium of Taiwan are projected to expand their distributions, whereas those species with medians higher than that of Taiwan are projected to contract upwards. Current distributional median elevation of a species, therefore, appears to be a new index for assessing species changing distributions with future warming.

Species with a current conservation status defined as common, uncommon, or rare based on the number of grids in which a species was seen, as defined by Ko et al. (2010), show a decreasing trend of occupancy in the climate-change scenarios (Table 2). For instance, by 2100 in both the A2 and B2 scenarios, the distributions of two common species, the White-whiskered Laughing Thrush (*Garrulax morrisonianus*) and the Collared Bush-Robin (*Tarsiger johnstoniae*), would occupy smaller distributional ranges than that of the Mikado Pheasant (*Syrmaticus mikado*), which is currently a rare species. According to the species distributional projections with gradually decreases, species that are now common may become rare with warming temperatures. Moreover, species used in this study are endemic, and being projected a decreasing distribution of these species, on the other hand, means simultaneously the threat of global extinction. Hence, we cannot ignore future ecological responses of common species to the effects of climate change.

The census data used in this study were collected throughout the island with a given location being censused 2–3 years on average. At some locations, however, there are longer term data sets. At these locations species distributional changes are occurring in the expected direction, *i.e.* upward shift in elevation. For instance, census data collected from 1992 to 2006 in Yushan National Park show that the richness of bird species increased at altitudes above 3500 m, because six montane species were found to have a higher upper distributional limit in 2006 than in 1992 (Lee 2008a, 2008b). The recent warming trend in Taiwan is likely to have affected the range boundaries of bird species directly by converting previously uninhabitable territories into habitable ones, and those effects are actually larger than what we predict in this study.

Beside, the impacts of climate change on bird distributions between islands and continents could be different and is valuable to be further explored. Recent findings on California breeding land birds showed anticipated coastward and upslope shifts in distribution in response to a warming climate (Wiens et al. 2009). We did not find any coastward patterns in Taiwan according to the inventory of avifauna since 1993. The main reason is due to different factors causing daily or yearly temperature changes between islands and continents. Critical factors influencing all of Taiwan, not just the coastal regions as sea-surface temperature would do on bigger islands like Australia, or coastal areas on continents, like coastal regions of California. Taiwan is small enough that the sea-surface temperature will influence the temperature throughout the island. Other reasons, for instance, (i) high human population densities occur in coastal areas with less vegetation, and (ii) the 17 species we used are montane bird species that prefer areas for high forest density are both partially influencing Taiwanese species. Importantly, as expected, our results were still consistent with "localized hotspots" (*i.e.*, a grid with high biodiversity) of change, especially in high-elevation areas.

Use of climatic scenarios

Ecological processes are usually gradual and can lead to continuous irreversible changes and evolutionary alterations. The four temporal snapshots in this study were selected to consider the possible reactions of species to a warming climate. Selecting several constant detection points (*i.e.*, years) to evaluate climate-change impacts on species distributions in general may encourage vigilance among ecologists and conservation scientists. We must remember, however, that there are other forces than climate change that are acting on species. These forces include habitat destruction, invasive species, over hunting and harvesting, for example. A sharp threshold had been estimate to exist between habitat loss-patch occupancy and climate change, which pointed that the habitat (loss or destruction) threshold would occur sooner during climate change and species would suffer more risks (Travis 2003). Invasive (non-native) species were found more successful to adapt to rapid variations in climate change by being provided a large range of idealistic conditions, which would further lead to reduce biodiversity or wipe out completely large areas of natural vegetation (Tausch 2008, Willis et al. 2010). The risk to species from climate change, therefore, may be much higher than that envisioned by our projections due to synergistic effects with these other stresses. Given the available data our results provided the best projections currently available. If, however, more data are available on dispersal ability, land-use change in various habitat locations, adaptations exhibited by all species, evolutionary changes along with other data, more robust results of species projected distributions can be obtained and inform actions in support of conservation and management.

Conclusions

Observed and projected distribution patterns show that the species studied are predicted to react differently to an increasingly warming climate, either increasing or decreasing their distributions. Our study demonstrated that current distributional median elevation of a species can be a new index for assessing species changing distributions in future warming with those having elevational medians below the elevational median of Taiwan would most likely have expanding distributions and those with medians above the available medians would most likely have contracting distributions. Indeed, endemic species deemed currently as common but have medians above the available medium could decrease their distribution so much that they become rare. Therefore, the possible future ecological response of current common species to climate change should not be ignored. Current common species could possibly have contraction of ranges, which could mean it might become a rare species when facing an increasing temperature. Therefore, understanding the possible effects of climate change on natural systems indeed provides more robust conservation and management practices to be determined by ecologists and governments. Species present distributional projections can be further strengthened with following long-term monitoring, targeted field-based observations and interdisciplinary experiments.

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Bird-monitoring in Europe – a first overview of practices, motivations and aims

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Abstract

Biodiversity monitoring is central to conservation biology, allowing the evaluation of the conservation status of species or the assessment of mechanisms of biodiversity change. Birds are the first taxonomic group to be used to build headline indicators of biodiversity due to their worldwide spatial and temporal coverage and their popularity. However, the landscape of bird-monitoring practices has never been characterized quantitatively. To objectively explore the strengths and weaknesses of the massive bird-monitoring effort in Europe we assessed the bird-monitoring practices, acquired with a questionnaire-based survey, in a sample of monitoring programs. We identify major correlates of among-program variability and compare monitoring practices from our database to recommendations of best monitoring practices. In total, we obtained responses from 144 bird-monitoring programs. We distinguish three types of monitoring programs according to the number of people that they involve: small, local-scale programs (56%), medium or regional programs (19%), and large-scale, national and international, programs (23%). In total, the programs in our sample involved 27941 persons, investing 79298 person days per year. Our survey illustrated that 65% of programs collected quantitative indices of abundance (count data). The monitoring design in a majority of the programs could be improved, notably in terms of unbiased spatial coverage, sampling effort optimization, replicated sampling to account for variations in detection probability, and more efficient statistical use of the data. We discuss the main avenues for improvement in bird-monitoring practices that emerge from this comparison of current practices and published methodological recommendations.

Keywords
conservation priorities, biodiversity monitoring, volunteers, survey design, conservation status in Europe, conservation policy support

Introduction

Biodiversity and environmental monitoring provide fundamental information for tracking environmental changes, to diagnose population trajectories and to provide conservation biology with relevant data. Such information is required for the design and evaluation of biodiversity policies, conservation management, land use decisions, and environmental protection. Biodiversity monitoring is therefore central to conservation biology, allowing the evaluation of the conservation status of species or to assess biological responses to environmental changes (such as climate change, Lepetz et al. 2009), and to conservation policy (Male and Bean 2005; Taylor et al. 2005; Donald et al. 2007).

A large number of monitoring programs have been developed and a large body of literature on biodiversity monitoring is available, including several articles that provide recommendations for an optimal design of monitoring programs (Danielsen et al. 2000; Yoccoz et al. 2001; Kery and Schmid 2004; Vořišek et al. 2008; Lindenmayer and Likens 2009; 2010). Apart from methodological advice, most of these articles agree that many monitoring programs were poorly designed and, therefore, could be a waste of time and resources (Nichols and Williams 2006). However, quantitative assessments of monitoring practices at varying spatial scales were not available at the time of these publications (Marsh and Trenham 2008). Large databases collecting data on and rating monitoring practices are now becoming available (Kull et al. 2008; Lengyel et al. 2008; Schmeller et al. 2009) and provide the first opportunity for a quantitative assessment of how well monitoring practices match methodological recommendations.

Bird-monitoring initiatives are the first provider of long-term monitoring data when institutional bodies set the goals of quantifying global biodiversity changes and of assessing the impact of environmental policies on biodiversity (Tucker and Heath 1994; Burfield et al. 2004; Gregory et al. 2005; 2006). In many instances, birds are the taxonomic group for which most data are available. Hence, we should characterize monitoring practices, and develop recommendations of how they could be improved for an optimized future monitoring effort. Further, such an assessment of the state of biodiversity-monitoring practices may contribute to the establishment of a global monitoring system, as envisaged by the Group of Earth Observation – Biodiversity Observation Network (GEOBON; Pereira et al. 2010).

For the first time, a comprehensive database of the FP6-project EUMON (hereafter DaEuMon; Schmeller et al. 2009) made available standard information describing biodiversity-monitoring practices in Europe. This meta-database contains data on sampling practices, sampling efforts, sampling design, volunteer involvement etc., of

600 European monitoring programs and aims at describing the monitoring landscape in Europe. Here we used this data source, to characterize bird-monitoring practices. We focused on differences among programs in motivation and aims, sampling design, sampling effort and methods of data analysis during the monitoring process. Further, we analyzed differences in these parameters among bird species groups (raptors, songbirds and near passerines, waterbirds), and according to the size of a monitoring program as defined by the number of persons involved. Our characterization of the overall European landscape of bird-monitoring practices will address the general questions: What are the average practices of bird-monitoring in Europe? And how do these practices relate to the motivation and aim for monitoring, to sampling effort, and to the involvement of non-professionals? A summary of this information will act as an aid for those wanting to launch a new program, improve the design of an ongoing monitoring program (adaptive monitoring), or evaluate bird-monitoring data quality. Our approach differs from earlier publications focusing only on national or international federations of monitoring programs (Gibbons 2000; Vořišek and Marchant 2003; Klvanová and Voríšek 2007) as we also include regional and local monitoring programs.

Methods

600 monitoring programs are available in DaEuMon database; They were obtained through a questionnaire survey (ESM1). Among them, 144 concern bird species and were analyzed in detail. We checked responses for completeness, and sought missing details from the coordinators of monitoring projects. Once the responses have been validated, data were made publicly available through our online database (http://eumon.ckff.si/biomat/). Complete information was not available for every single question for all programs, hence affecting sample sizes in the analyses.

For the characterization of the bird-monitoring landscape, we focused on differences in the motivation and aims, sampling design, sampling effort and methods used for data analysis. We analyzed the differences between bird species groups (raptors, songbirds and near passerines, waterbirds) and between monitoring programs of different sizes in terms of the number of persons involved. We defined three size-categories: small ($N_{persons} \leq 30$; N = 81), medium ($N_{persons}$ 31 – 150; N = 26) and large monitoring programs ($N_{persons} > 150$; N = 32). The motivation was characterized by the program objective (scientific, management or political/juridical), the type of trends monitored (distribution, population size or avian community trends) and the focal ecological factor (climate change, habitat fragmentation, pollution, invasive species, land use). Sampling design was characterized by site choice methodology, the use of stratified sampling or not, the use of repeated sampling or not (that allow accounting for detection probability), the location of sampling sites within and/or outside protected areas, and the main field data type collected (Presence/absence, Counts, Mark recapture, Age/size structure, Phenology). We further quantified the sampling effort by the number of species ($N_{species}$), persons ($N_{persons}$), sites (N_{sites}), visits per site (N_{visits}), sampling effort in person.days, and the proportion of volunteers (%Vol).

We tested for differences in practices with generalized linear models (GLM) using SAS 9.1.3 (Cary, USA, 2002); GLM with a multinomial distribution of error terms and a clogit link function for the type of field data (categorical variable); GLM with a Poisson distribution of error terms and a log link function for the number of species monitored; GLM with a binomial distribution of error terms and a logit link function for the analysis of the use of stratification, of detection probability, and of advanced statistics. The dependent variables were therefore: the type of field data, the number of species monitored, the use of stratification, the use of detection probability, and the use of advanced statistics. The corresponding independent variables included in the models were: the number of persons involved in the program, the number of professionals, the ratio of volunteers, the number of person days, and the program objective. We also included the sampling design used when analyzing the use of advanced statistics. The models were adjusted for overdispersion when necessary. We conducted a stepwise procedure with a backward elimination at the 5%-level, starting with a fully saturated model, incorporating all independent variables with no interaction, and dropping, step-by-step, all non-significant variables. At each step, the term that gave the smallest contribution to the model (largest p-value) was excluded.

Bias in geographic coverage

A major problem of surveys such as ours (volunteer response to a mailed questionnaire) is that it is nearly impossible to achieve a random sample because of the decentralized structure of the network of monitoring activities (Schmeller et al. 2009 for Europe and Marsh and Trenham 2008 for North America). Indeed, monitoring coordinators of highly visible monitoring programs have a certain fatigue toward questionnaires or strong time constraints and simply may not reply (Barclay et al. 2002). At the opposite end of the size gradient, it is hard to get in touch with a large number of local, non-federated monitoring programs, which represent a large subset of the available monitoring data. The EuMon survey encountered both problems. For example, not all coordinators of national Breeding Bird Surveys (listed on the page of the Pan-European Common Bird-monitoring Program, PEBCM; EBCC 2010) contributed to the EuMon survey. A direct comparison of programs covered by both surveys is difficult, as the EuMon survey covered a much larger range of different monitoring programs (from local to international) than the EBCC list, which focuses on national programs only. Further, titles of national programs differed between the EBCC and EuMon surveys.

Despite a large effort in sending out requests for cooperation to a wide audience, our survey data provide a characterization of monitoring practices in Europe that suffers from a biased geographic coverage. We used GoogleScholar to estimate the bias in our sample by looking for articles with the search string ("bird-monitoring" OR "bird survey" country). Our analysis shows that Lithuania, Poland, France, Bulgaria and Andorra were overrepresented in our program, while Great Britain, for example, was underrepresented (Figure 1). Also in comparison to data collected by EBCC, our survey has obviously undersampled bird-monitoring programs in Great Britain and Sweden. Our survey covers 24 European countries, with a strong (over-) representation of France and Poland (Figure 1; Schmeller et al. 2009). Despite this non-random coverage of European countries, our database is for now the most extensive data set to characterize bird-monitoring practices in Europe. Other initiatives analyzing bird monitoring programs focused on large-scale, national breeding bird surveys (Gibbons 2000; Vořišek and Marchant 2003; Klvanová and Voríšek 2007), which may be considered as the most visible and legitimate minority within the whole bird-monitoring community.

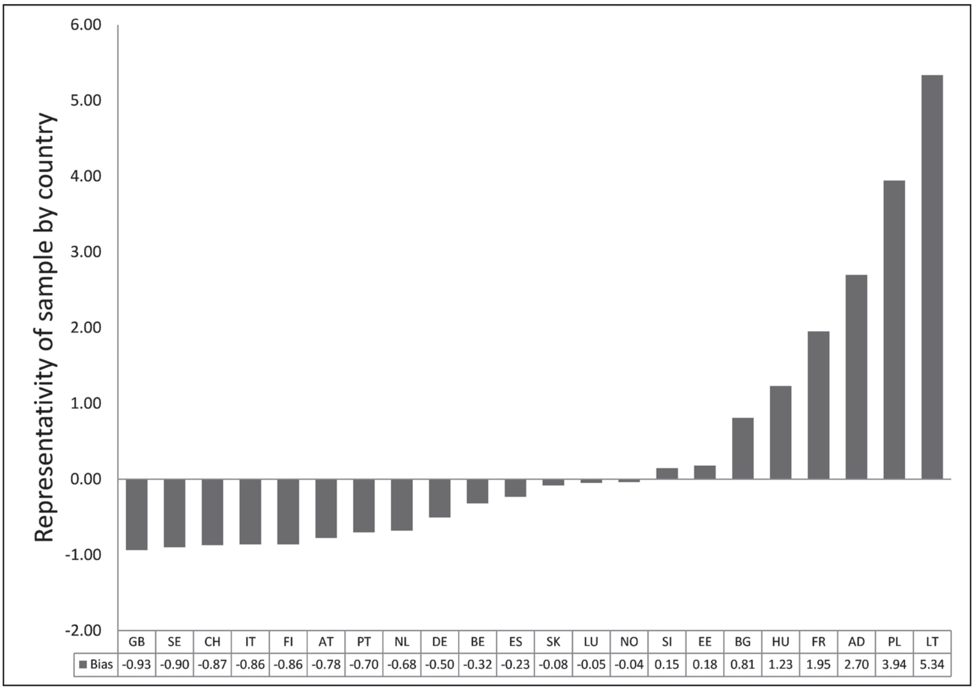

Figure 1. Estimation of the bias in the number of bird-monitoring programs in the EuMon database per country (bias = [Number of programs DaEuMon – Number of articles in Google Scholar]/ Number of articles in Google Scholar). The reference to quantify bird monitoring activity per country was the number of publications in GoogleScholar returned for the search string ("bird-monitoring" OR "bird survey" AND country name). The countries are abbreviated following the two-letter convention of the international community (ISO 3166-1 alpha-2 codes; GB = Great Britain, SE = Sweden, CH = Switzerland, IT = Italy, FI = Finland, AT = Austria, PT = Portugal, NL = Netherlands, DE = Germany, BE = Belgium, ES = Spain, SK = Slovakia, LU = Luxembourg, NO = Norway, SI = Slovenia, EE = Estonia, BG = Bulgaria, HU = Hungary, FR = France, AD = Andorra, PL = Poland, LT = Lithuania).

Results

Our European-wide survey yielded responses from 144 bird-monitoring programs employing 27941 persons investing 79298 person days per year. The majority of responses recorded in our database came from France (49; 34%) Poland (28; 19%), and Lithuania (13; 9%). Six to eight responses came from the Netherlands, Germany, Spain, Norway, and Hungary (35; 24%; Figure 2). In total, all bird-monitoring programs employed 27941 persons, with a mean of 201+/- 75 persons per program and a mean manpower of 615 +/- 138 person days per year per program.

Small programs monitored 29 +/- 5.2 species (median = 6) with 11 +/- 1 persons (median = 7), which were mainly professionals (66% +/- 4.5), investing on average 148 +/- 34.2 days per year, visiting 116 +/- 67.7 sites (median = 12) on average 9.4+/-2.9 times (median = 3). Medium sized programs monitored 42 +/- 17.8 species (median = 5) with 64 +/- 4.7 persons (median = 60), which most frequently were volunteers (77% +/- 5.7), investing on average 492 +/- 109 days per year, visiting 67 +/- 16.9 sites (median = 43) on average 28.7+/-19.1 times (median = 9.5). Large programs monitored 72 +/- 12.2 species (median = 70.5) with 793 +/- 306 persons, of which most

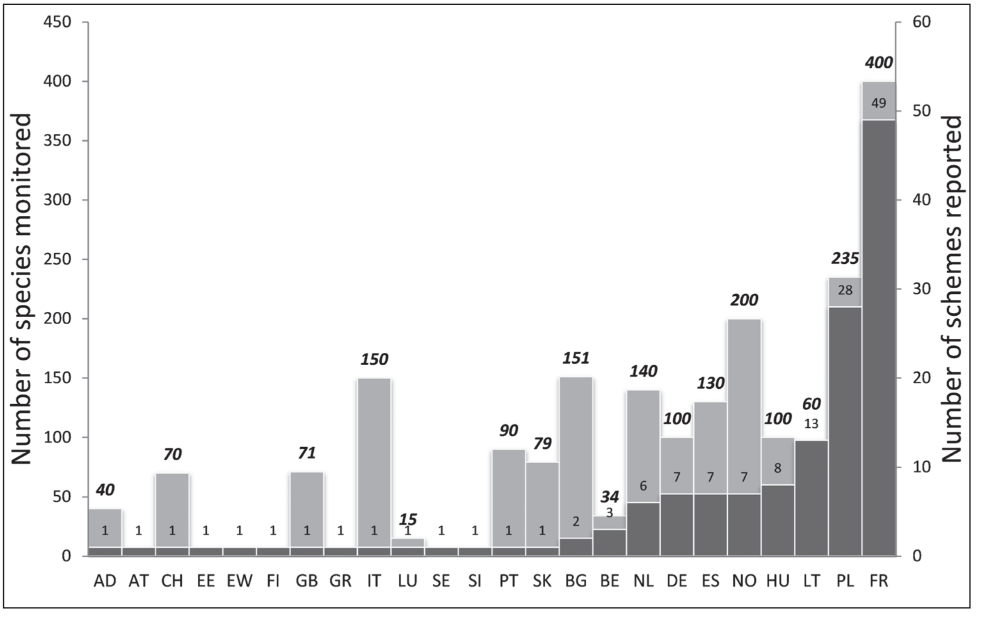

Figure 2. Number of responses (dark grey) and number of species monitored (light grey) per country. The countries are abbreviated following the two-letter convention of the international community (ISO 3166-1 alpha-2 codes). (AD = Andorra, AT = Austria, CH = Switzerland, EE/EW = Estonia, FI = Finland, GB = Great Britain, GR = Greece, IT = Italy, LU = Luxembourg, SE = Sweden, SI = Slovenia, PT = Portugal, SK = Slovakia, BG = Bulgaria, BE = Belgium, NL = Netherlands, DE = Germany, ES = Spain, NO = Norway, HU = Hungary, LT = Lithuania, PL = Poland, FR = France). Bold italic values correspond to the number of species.

are volunteers (82.3% +/- 5.8), investing on average 1939.6 +/- 534.6 days per year, visiting 1098.7 +/- 357.1 sites (median = 560) on average 7.3+/-3.7 times (median = 2; see Figure 3 for more details).

The main factors of ecological change that coordinators considered that they could assess with their monitoring data were land use change in small and medium programs (Table 1). In large programs, a majority of programs monitored land use changes and climate change impacts (Table 1). The distributions of the ecological factors monitored differed significantly between the differently sized monitoring programs ($\chi^2_2 = 6.879$, $p = 0.032$; Table 1). In all three categories of monitoring programs, population trends were the first target of the monitoring. Community trends were least monitored across all programs sizes (Table 1). Most of the small and medium programs were scientific programs. In medium programs, many also had a management motivation (34.6%), while the large programs included 34.4% scientific, 28.1% political and 25% management programs ($\chi^2_2 = 1.294$; $p = 0.523$; Table 1).

In small programs, sites were mainly chosen through expert knowledge (Table 1). In medium programs, sampling was most frequently exhaustive or based on site choice according to expert knowledge (Table 1). In large programs, random sampling and site choice by expert knowledge was most frequently employed (Table 1). Whether monitoring was conducted within and/or outside a protected area was independent of monitoring program size, as was the field data type that were most frequently used (Table 1). The issue of detection probability was neglected in all types of programs ($\chi^2_2 = 0.092$; $p = 0.955$), ranging from 35% of the total number of small programs to 46.7% (large programs). The same result was found for the application of stratified sampling ($\chi^2_2 = 2.656$; $p = 0.265$; Table 1). In small and medium programs, basic statistics (descriptive statistics or correlations) were most frequently used, while large programs may have more frequently used more advanced statistics ($\chi^2_2 = 3.348$; $p = 0.188$).

Several monitoring programs focused on one of the three bird species groups, raptors (N = 16), waterbirds (N = 44), and songbirds and near passerines only (N = 27). Raptor programs monitored 4.8 +/- 1.3 species (median = 3) with 42.6 +/- 16.3 persons (median = 11), which were mainly professionals (58% +/- 11.6; median 14.3%), investing 392.5 +/- 127.9 days per year, visiting 164.1 +/- 91.5 sites (median = 23) on 3.7 +/- 1.0 times (median = 2). The majority of raptor programs did not consider detection probability nor stratification (87.5%), but followed an exhaustive sampling design (56%) and analyzed data with basic statistics only (12.5%). Waterbird programs monitored 16.9 +/- 4.1 species (median = 1) with 138.9 +/- 52.6 persons (median = 30), which were mainly volunteers (52% +/- 6.2; median 60.0%), investing on average 869.4 +/- 374.5 days per year, visiting 294.3 +/- 111.1 sites (median = 50) on average 22.3 +/- 12.1 times (median = 2). In waterbird programs 25% considered detection probability and 20.5% stratified their sample. The sampling design was either exhaustive (36.4%) or following expert knowledge (47.7%), while only 9% employed random or systematic sampling. Songbirds and near passerine programs monitored 26.0 +/- 8.0 species (median = 12) with 400.7 +/- 369.3 persons (median = 22), which were mainly volunteers (59.8% +/- 8.4; median 87%), investing on average 222.2 +/- 72.5 days per year, visiting 409.3 +/-

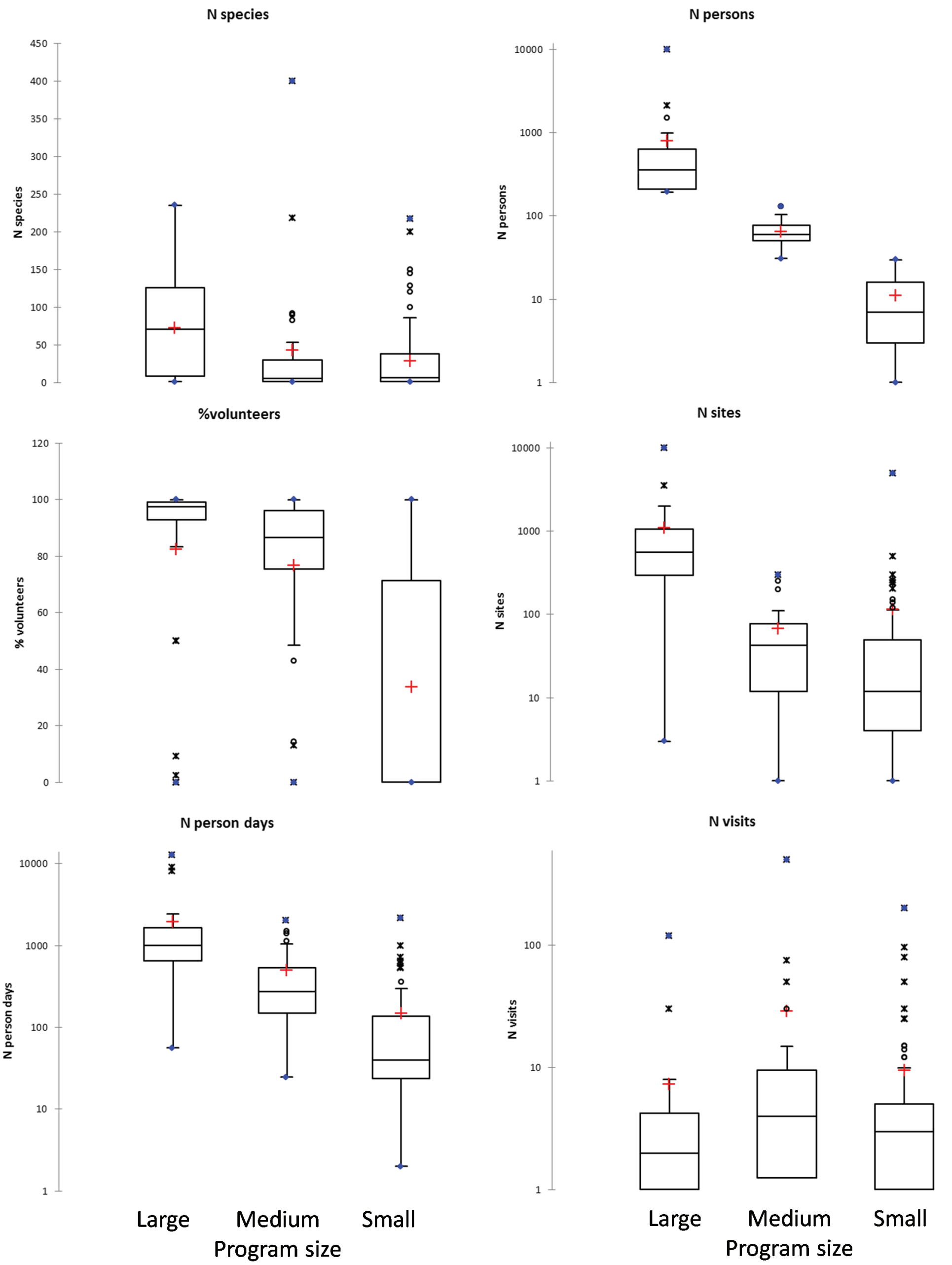

Figure 3. Univariate boxplots on the sampling effort and proportion of volunteers for small, medium and large European bird-monitoring programs (the size of a monitoring scheme was defined mainly by the number of people involved).

Table 1. Summary of responses of European bird species programs to questions on the motivation and aims, and the sampling design. The values are given for small, medium and large programs as defined by the number of people involved in the monitoring (see also text). For more information on the questions, see the supplementary material ESM 1, which is available online.

		Small N (%)	Medium N (%)	Large N (%)
Motivation and aim				
	causes of change a program monitors			
	Land use	46 (56.8%)	15 (57.7%)	20 (62.5%)
	Climate change	20 (24.7%)	11 (42.3%)	17 (53.1%)
	Habitat fragmentation	19 (23.5%)	3 (11.5%)	8 (25.0%)
	Pollution	18 (22.2%)	3 (11.5%)	9 (28.1%)
	Invasive species	9 (11.1%)	4 (15.4%)	8 (25.0%)
	Reason for launching a program			
	Scientific	33 (40.7%)	11 (42.3%)	11 (34.4%)
	Political	24 (29.6%)	4 (15.4%)	9 (28.1%)
	Management or restoration	21 (25.9%)	9 (34.6%)	8 (25.0%)
	Other	2 (2.5%)	2 (7.7%)	4 (12.5%)
	Monitored trends			
	Population trends	79 (97.5%)	26 (100%)	31 (96.9%)
	Distribution trends	44 (54.3%)	16 (61.5%)	25 (78.1%)
	Community trends	41 (50.6%)	5 (19.2%)	7 (21.9%)
Sampling design				
	Choice of site			
	Personal or expert knowledge	46 (65.0%)	9 (37.5%)	12 (37.5%)
	Exhaustive sampling	26 (32.5%)	10 (41.7%)	4 (12.5%)
	Systematic sampling	4 (5.0%)	2 (8.3%)	2 (6.3%)
	Other	3 (3.7%)	3 (12.5%)	5 (15.6%)
	Random sampling	1 (1.3%)	-	9 (28.1%)
	Field data type			
	Age	-	-	1 (3.1%)
	Counts	53 (65.4%)	14 (53.8%)	24 (75.0%)
	Mark-recapture	16 (19.7%)	7 (26.9%)	2 (6.3%)
	Presence-absence	8 (9.9%)	3 (11.5%)	2 (6.3%)
	Phenology	4 (4.9%)	2 (7.7%)	3 (9.4%)
	Monitored area legally protected			
	Both, within and outside a reserve site	49 (61.3%)	17 (65.4%)	28 (87.5%)
	Within reserve site	21 (26.3%)	3 (11.5%)	-
	Outside reserve site	10 (12.5%)	6 (23.1%)	4 (12.5%)
Data processing				
	Detection probability	28 (35.0%)	9 (36.0%)	14 (46.7%)
	Stratification	18 (22.2%)	8 (30.7%)	12 (37.5%)

383.7 sites (median = 10) on average 11.8 +/- 3.4 times (median = 2). Songbirds and near passerine programs accounted for detection probability in 63% and used stratification in 26% of the programs. Most of the programs (74.1%) used a sampling design following expert knowledge and used advanced statistics (63%) for data analysis.

Sampling and data processing

The field data type largely depended on the program objective (χ^2_2 = 10.11; p = 0.006): programs with a scientific motivation more frequently employed mark-recapture studies (35%) as compared to management/restoration programs (10.5%), while politically motivated programs did not employ mark-recapture methods at all. Conversely, counts were used less frequently in scientific programs (46%) as compared to management/restoration programs (71%) and programs with a political interest (84%).

Site choice methodology was related to the proportion of volunteers involved (χ^2_1 = 4.67; p = 0.031). Programs with more professionals than volunteers employed systematic sampling or chose sites based on expert knowledge, while programs with exhaustive or random sampling were dominated by volunteers. Consideration of detection probability was related to the program objective (χ^2_2 = 16.71; p < 0.001): scientifically oriented programs accounted more often for detectability than other programs, although still 46% of the scientifically motivated programs ignored the problem of detection probability as did 66.7% for management programs and 82.8% for political programs. Stratification was used in few programs (31% of scientific programs; 23.7% of management programs; 16.2% of politically motivated programs; χ^2_2 = 2.043; p = 0.36).

Advanced statistics (i.e. GLM, or Generalized Additive Models) were more likely used for data analysis with increasing total sampling effort (number of person days) and varied with the program objective (respectively, χ^2_1 = 11.58; p < 0.001; χ^2_2 = 14.76; p < 0.001); 62.5% of the scientific programs used advanced statistics, 47% in management programs, and 23.5% in politically motivated programs. The level of statistical data processing (use of basic or advanced statistics) was not related to the sampling design (χ^2_4 = 6.04; p = 0.196).

Discussion

The majority of programs of our database comprised of small programs, i.e. monitoring few bird species with few people. These programs were homogeneous in terms of practices for monitoring bird populations on a local scale using counts or even capture-mark-recapture data to monitor population trends in detail. Capture-mark-recapture data were usually collected in scientifically motivated programs at sites chosen by experts. Fewer programs were medium-sized, focusing on populations on a local to regional scale, using count data and an exhaustive sampling design. The large monitoring programs sampled count data, while selecting sites either randomly or following expert opinion.

Monitoring programs share the common desire to determine what changes are occurring in bird populations and why these changes occur. Programs at different scales are needed to address these questions, although their primary aims may differ depending on the scale of implementation. Large-scale monitoring programs across biogeographic regions, countries or a continent are usually designed to determine if population changes are occurring. However, the design of large-scale programs is too coarse to provide information on changes at specific sites or to provide direct information on the causes of population change. Here, small-scale monitoring programs are needed to analyze why population change is occurring at specific sites. Such local-scale data can then feed into management and conservation actions for specific sites. With these differences in mind, it is little surprising that population trends are by far the most frequently monitored trend, regardless of the size of the monitoring program.

Due to the aims of a local scientific program, few employed random sampling, while site selection was done according to expert knowledge. While such a design is suitable for specific (scientific) questions, a subjective sampling effort in general must be considered as a poor design for a monitoring program since it provides a biased coverage of the mechanisms at play, without characterizing the biases. Surprisingly, our data suggest that random sampling, while highly recommended, was employed by only 28% of the large-scale programs and hence 72% did not follow the recommendations of good monitoring practices (Gaines et al. 1999; Yoccoz et al. 2001; Nichols and Williams 2006). Note that the national Breeding Bird Surveys (Gibbons 2000) usually employ randomized or semi-randomized sampling but large-scale, national bird-monitoring programs formed a minority of the schemes in our database.

Concerning data collection, bird-monitoring data were usually counts, largely dominating across all monitoring programs in our database. Resource intensive capture-mark-recapture studies (Voříšek et al., 2008) were usually conducted at local and regional scales. The small and locally focused monitoring programs, however, need to be put into a large-scale perspective to determine if changes are due to local or external factors. Such a consideration is important for a generalization of trends across geographic and temporal scales. Therefore, it is important that the results from small monitoring programs are interpreted relative to changes at the population level. They can then serve as benchmark sites for large-scale monitoring programs, thereby providing in-depth information at specific sites (Downes et al. 2005; Henry et al. 2008). Our analysis shows that the potential of such an integration of local and small monitoring programs on a larger-scale is high, given that the homogeneity of the different parameters analyzed in our sample of small monitoring programs was comparably high. Integrating the monitoring data of the 81 small monitoring programs could yield a remarkably good coverage and profound insight of local impacts on bird populations across Europe.

In respect to the determination of the causes of change in population trends, it is also important to monitor sites in and outside of protected areas since the pressures are different. Our data suggests that this notion is well implemented in bird-monitoring in Europe, improving our ability to generalize results by comparing population changes within and outside of protected areas. Such comparisons are of special importance to disentangle large-scale factors (such as climate change), from more local effects (such as habitat fragmentation and pollution for instance).

Concerning sampling stratification, we also found a difference between the differently sized programs, which is likely to be related to the differences in the aim and design of small to large programs. In small programs, stratified sampling was applied in only 22.2%, while in large programs the proportion raised to 37.5% (30.7% in medium programs). For local and regional programs such a proportion might be sufficient since homogeneity of the sample population is higher at a smaller scale. In contrast, stratified sampling must be employed more frequently in large-scale programs due to limited resource and sampling disequilibrium between potential strata.

The largest deficit in the consideration of recommendations was the lack of repeated sampling to account for detection probability. Only little more than a third of programs employed repeated sampling, usually programs with a scientific motivation. Programs with management objectives and with a political motivation employed repeated sampling even less often, making them more prone to misinterpretation of trends that may be due to variations in detection probability (Pellet and Schmidt 2005; Schmidt 2005; Kery et al. 2006). Here, large programs performed the best. In programs on a local scale, detection probability might not be considered due to two reasons: (i) detection of a focal species is considered sufficient as sites are visited more frequently or enough so that trends are not biased (but see Archaux et al. 2011), or (ii) the statistical analysis needed to model detection probability appears too complex. This would be coherent with the fact that small and medium programs usually employed only basic statistics, while large programs used advanced statistics to value their large datasets. Hence, recommendations of good monitoring practice are only followed by a minority of the programs, with many consequences for the interpretation of data, especially in politically motivated programs.

Generally, our data show that there is a huge variety of monitoring practices across all monitoring programs, among and within bird species groups, partly explained by the program objective, and the scale of the implementation of programs. It appears to be justified to recommend that bird-monitoring in Europe may step up the effort in methodological implementation of monitoring recommendations (Vořišek et al. 2008) to produce more standardized bird-monitoring data. Such an effort would increase the potential uses of these data, and particularly the potential for the integration of data at large geographical scales (Downes et al. 2005; Henry et al. 2008; Pereira et al. 2010).

Resource limitations and volunteer-based monitoring

The culture of bird-monitoring was born and propagated by visionary bird watching and naturalist amateurs, led by skilled professionals. This enabled the founding of long-term databases with minimal funding. Due to this historical contingency, the involvement of volunteers in monitoring is still key to maximize the sampling effort and to acquire a large-scale image of changes in bird diversity (Engel and Voshell Jr. 2002;

Bell et al. 2008; Schmeller et al. 2009). At first sight, our survey concurs with the common belief that an optimized sampling design is poorly compatible with massive volunteer involvement. The recommended stratified and/or random spatial sampling (Yoccoz et al. 2001; Voříšek et al. 2008) is used in only a rather small proportion of the monitoring programs, suggesting that coordinators believe that they cannot impose sophisticated sampling designs if they want to attract large numbers of volunteers. However, our survey shows that 14% of programs have successfully used random sampling designs, further improved by the use of advanced statistical analysis, showing that volunteer involvement is actually compatible with good monitoring practices (Schmeller et al. 2009). This optimization of monitoring constraints is well illustrated by Voříšek and colleagues (2003; 2008), in their overview of the national common bird surveys in Europe (http://www.ebcc.info/pecbm.html). Random sampling could be achieved in most of the programs once volunteers see the advantages of random *versus* opportunistic sampling (Buckland et al. 2005). We believe that the key to improving average monitoring practices is the involvement of skilled biologists, engaging in training and effective communication regarding sampling design and data processing and analysis.

Recommendations

In the monitoring literature a three-phase approach is described for the process of biodiversity monitoring, (i) identifying monitoring questions and aims, (ii) identifying the most suitable monitoring methods, and (iii) interpreting monitoring data (Gaines et al. 1999; Yoccoz et al. 2001; Voříšek et al. 2008). For most monitoring programs, the best data type to be collected is count data, which enable management actions and secure an early warning for conservation and policy. More sophisticated methods like capture-mark-recapture studies can then be employed to explore more specific scientific questions (Lepetz et al. 2009). Count data has the best trade-off between resource use for data collection and the quantity of information contained in the data. Further, monitoring could be stratified to optimize resource allocation between independent samples (i.e., sites), and employ random (or systematic) sampling to secure an unbiased spatial coverage. Importantly, detection probability needs to be accounted for since even low differences in detection probability between site or years can induce spurious conclusions (Archaux et al. 2011). It means that repetitive sampling of the same sites within a year should be the rule. In case of limited manpower, Voříšek and colleagues (2008), among others, recommended to maximize the number of samples, even at the expense of the size of each sample, so that the precision of population estimates remains the highest possible, allowing a better coverage of the different sources of heterogeneity which in turn can also limit the bias. For the statistical analysis, it is advantageous to not only use descriptive statistics or simple correlation analysis, as these techniques do not optimize the extraction of the information contained in the monitoring data. There is a range of different free software packages available, which could be used to do advanced statistics with count and capture-mark-recapture data (e.g. TRIM, MARK, and several R packages). For further valorization of monitoring data, coordinators need to consider data integration across different monitoring programs. Therefore, guidelines for data integration across programs need to be clear, comprehensible and accessible to monitoring coordinators (Henry et al., 2008). Further, more collaborations between monitoring programs at different scales need to be established, so that the numerous datasets currently not included in the evaluation of trends in bird populations might be better considered in the future. Finally, monitoring coordinators may wonder how to attract volunteer monitors for a specific program to increase the manpower without over-stretching the financial budget. Several factors define a successful volunteer involvement (Bell et al. 2008; Schmeller et al. 2009; Vandzinskaite et al. 2010): (i) the socio-political background influences levels of participation, (ii) different recruitment strategies are needed for retention of volunteers, (iii) keep volunteers informed, (iv) carefully consider relationships between professionals and volunteers, and (v) collaborate with other monitoring programs to add value.

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Appendix

Supplementary material to article European Bird-monitoring - an overview. (doi: 10.3897/natureconservation.2.3644.app) File format: MS Word Document (doc).

Explanation note: The questionnaire was designed to assess how biodiversity monitoring schemes were carried out and what the motivation was to launch that scheme.

Note: The following is a transcript of the questions coordinators answered in the online questionnaire available at: http://eumon.ckff.si/monitoring. The online version also contains clarifications and explanatory notes.

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