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SPECIAL FEATURE: PERSPECTIVE PAPER

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Filling the gaps in ecological studies of socioecological systems

Abstract Our human-dominant world can be viewed as
being built up in two parts, social and ecological sys-
tems, each consisting of multi-level organizations that
interact in a complex manner. However, there are
knowledge gaps among those interactions. In this paper,
we focus on studies filling two types of gaps in the
socioecological system, some of which are case studies in
the East Asia region and others are discussed in a more
general context. First, we address the gaps between
different levels of organizations in ecological systems,
namely, (1) the importance of plant trait plasticity in
bridging evolution and ecology, (2) linking primary
producer diversity and the dynamics of blue carbon in
coastal ecosystems in the Asia–Pacific region, and (3)

research direction of climate change biology to fill the
gaps across evolution, community, and ecosystem. Also
included is (4) the gap between ecological monitoring
programs and theories, which also addresses the poten-
tial of citizen science. Second, we illustrate the gaps
between ecological and social systems through ongoing
development of an ecosystem management framework,
i.e., ecosystem-based disaster risk reduction. Finally, we
summarize the benefits of filling the gaps for ecologists
and society.

Keywords Blue carbon · Ecosystem-based disaster risk
reduction (ECO-DRR) · Citizen science · Climate
change-induced redistribution · Plant plasticity

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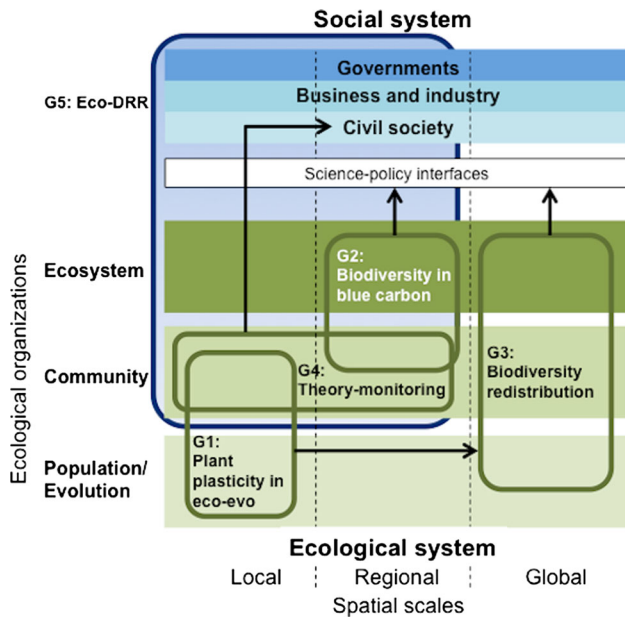
Fig. 1 Map to link ecological and social systems and filling gaps within/between two systems. In this map, we address different levels of organization, especially for ecological systems. G1–G5 correspond to the section "Filling the Gap 1" through "Filling the Gap 5" in the text. Arrows imply potential links that could fill the gaps further but are not fully discussed in this article. Lead authors of these five sections are T. Ougushi, H-J Lin, I-C Chen, J Urabe, and T Yoshida, respectively

Introduction

Our human-dominant world involves two interacting parts, social and ecological systems (Chapin et al. 2009). In the social system, different organizations (citizens, scientists, civil society, business and industry, and governments) interact, resulting in diverse human actions. The ecological system has a hierarchy of diverse entities and processes (e.g., evolution, population, community, and ecosystem) among levels of biological organization. Both systems have various spatial and temporal scales. The two subsystems interact; human actions impact ecological systems, whose responses in turn affect human actions, forming a socioecological feedback system. To maintain sustainability of the earth ecosystem, research from a broad perspective is required in future ecology study (Chapin et al. 2010; Future Earth 2013; Itoh et al. 2017).

To develop a framework based on such a broad perspective, it is necessary to construct a map that highlights the interrelationships among components in the socioecological system. At present, there are many gaps in the map, owing to diversification of science and gaps between society and science. To bring the map closer to completion, it is necessary to point out where the major gaps are located in order that they will be bridged (Agrawal et al. 2007; Enoki et al. 2014; Nakadai 2017).

This paper is the one of the achievements of the 5th Japan-Taiwan Ecological workshop entitled "Filling the Gaps: What's missing in the genotype–phenotype-population-ecosystem continuum?", which was held in November 2016 at Ryukoku University, Kyoto, Japan (see details in Nakamura et al. in this volume). The authors of the present paper formed part of the organizers and invited speakers of this workshop. These authors, with their diverse research backgrounds, selected some of the major gaps in ecological and interdisciplinary studies of socioecological systems. The selected topics include not only general research questions in ecology but also case studies of ecological or socioecological systems in Taiwan and Japan (Fig. 1).

In this paper, we focus on two types of gaps in the study of socioecological systems (Fig. 1). In East Asia, there is substantial biodiversity (Nakano et al. 2012), a strong contribution to the global biogeochemical cycle (Tian et al. 2013), large populations, and frequent natural disturbances (Wu et al. 2005).

The first gaps addressed are those between different organization levels or disciplines that occur across wide spatial scales in ecology, for which we introduce four specific research topics as examples. The first topic addresses a gap between different disciplines, evolutionary and ecological research, which is found in studies of plant–herbivore communities. We discuss the possibility that this gap may be filled by considering plant trait plasticity (Gap 1). The second topic is the knowledge gap between primary producer diversity and the dynamics of "blue carbon" in coastal ecosystems and its impacts on the carbon budget, especially in the Asia–Pacific region (Gap 2). The third topic regards the climate change-induced redistribution of organisms. Recent advances in climate change biology have improved understanding of species and biodiversity redistribution, but there remains a gap to be filled between the redistribution, community reorganization, and renewed ecosystem processes of individual species (Gap 3). Research achievements and knowledge obtained by filling Gaps 2 and 3 will contribute to bridging science–policy interfaces such as the Intergovernmental Panel on Climate Change (IPCC) and Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services (IPBES) and environmental management within local and regional governments. The last topic treats the importance of ecological monitoring programs in advancing ecological theories through the example of the monitoring program following the 2011 Tohoku Earthquake tsunamis. This example also highlights the gap between two of the basic individual units of society, ecologists (scientists) and citizens (Gap 4) and the potential of citizen science.

Second, we address the gaps between ecological and social studies. Specifically, we discuss the ongoing development of a new ecosystem management framework in Japan, ecosystem-based disaster risk reduction (ECO-DRR), as a tool to fill the gap between ecological science and traditional engineering-based management on various scales of governments (Gap 5). After illustrating examples of these two types of gaps, we then indicate the benefits and next challenges toward completing the map of coupled socioecological systems.

Filling the gaps

Filling the gap 1: The major role of plant trait plasticity in eco-evo feedbacks between plant and associated herbivore communities

Because there are multiple levels of ecological organization from genes to ecosystems, ecologists have long recognized the importance of integrating across levels. The interplay of evolution and ecology is critical for understanding the evolution of biodiversity, community structure, and ecosystem functions (Schoener 2011). Recent studies of plant–herbivore interactions have highlighted that genetic diversity in plants can be a driver of plant-based communities of herbivores (Whitham et al. 2006), because plant genotypes produce phenotypic variation. However, herbivory can also produce phenotypic variation of plants (i.e., herbivore-induced phenotypic plasticity). Although phenotypic and genetic variation among individual plants is critical for the interplay of ecology and evolution, current research into eco-evolutionary dynamics lacks explicit consideration of phenotypic plasticity as a source of phenotypic variation.

We outline a conceptual framework of eco-evolutionary dynamics of plant–herbivore systems (Fig. 2; Ohgushi 2016). There are two sources of the phenotypic variation in plants, genetic variation and phenotypic
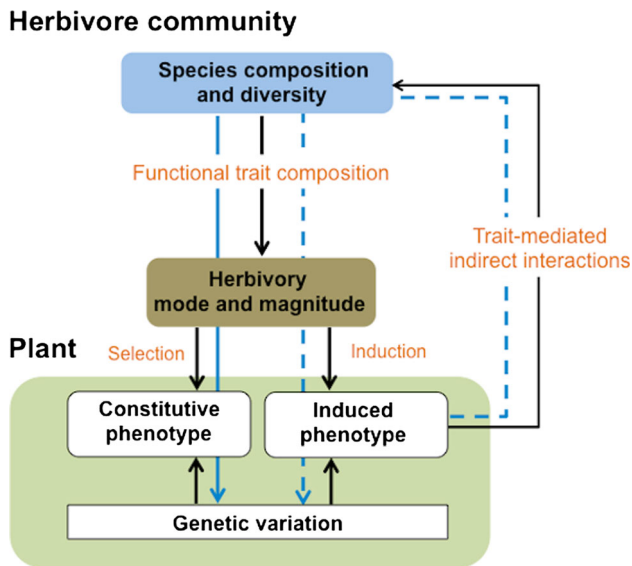
Fig. 2 Roles of plant trait plasticity in eco-evo feedbacks between plant and herbivore community (Gap 1). Species composition and diversity of herbivore community determine the type and magnitude of herbivory through functional trait composition, which can directly act as selection pressure on plant constitutive phenotype (direction selection, blue solid line). At the same time, herbivory properties only induce the phenotypic changes of plant traits, which first act as selection pressure on herbivore traits, which in turn feed back as selection pressure on plant traits (indirect selection, blue-dashed line). Figure is modified from Ohgushi (2016)

plasticity, both of which are greatly influenced by herbivory. Herbivory imposes selection on plant genotypes and induces plastic trait changes. Because induced plant phenotypes have a genetic basis (Gols et al. 2008), plant trait evolution may affect the likelihood and magnitude of phenotypic plasticity, which in turn affects the mode of adaptive evolution (Ghalambor et al. 2015). Increased variation in phenotypic plasticity of plants enhances trait-mediated indirect interactions among herbivores, and thereby links direct/indirect and trophic/non-trophic interactions (Ohgushi 2005). This can alter species composition and diversity of herbivores. Regarding feedback from herbivore communities to plant trait evolution, those communities can increase or decrease herbivory. This is because species and functional diversity of herbivores that induce plant phenotypic plasticity increase or decrease abundances of herbivores, resulting in variable intensities of overall herbivory. Species and function of these inducers can influence the herbivory intensity in different ways. These changes in abundances of herbivore community members determine not only the strength of selection on plant traits but also the expression of induced phenotypes. There are two pathways—direct and indirect—of selection by herbivore communities for plant traits. Direct selection can occur when herbivore community properties select plant traits. In contrast, indirect selection can occur when herbivore communities induce plant phenotypes, and these induced phenotypes feed back to exert selection on herbivore traits, which in turn lead to further selection of plant traits.

Research on plant–herbivore interactions may offer promising prospects to stimulate research on eco-evolutionary dynamics by incorporating induced plant phenotypes. First, herbivore-induced plant phenotypic plasticity is very common and widespread in nature (Karban and Baldwin 1997), and is an essential source of plant phenotypic variation. Second, induced plant phenotypes can strongly influence composition and diversity of a plant-based community of herbivores (Ohgushi et al. 2007). Third, plants support a wide range of herbivores with different guilds, which induce plant phenotypes in different ways.

There are some critical needs to further develop eco-evolutionary perspectives in plant–herbivore communities, as follows. (1) Since community and ecosystem consequences of plant genetic diversity have been well documented recently, it is essential to understand the importance of induced phenotypes of plants relative to genetic diversity, and their interactive effects on herbivore community structure and plant trait evolution (Hanski 2012). (2) Because selection pressure imposed by multiple herbivores may be non-additive, we must explore species and functional diversity and community herbivore composition to understand selective regimes not only for plant traits but also for herbivore host-use traits. (3) We should appreciate the indirect selection of the herbivore community of plant trait evolution via evolution of host-use traits of herbivores, in response to

plant phenotypic plasticity. (4) Because the structure of plant-based communities of herbivores varies across spatial scales (Ohgushi and Hambäck 2015), we must understand how the geographic structure of those communities shapes the interplay of ecological and evolutionary dynamics (Thompson 2013). Research into eco-evolutionary dynamics of plant–herbivore communities will undoubtedly enrich understanding of a broad scope of evolution and ecology of plant–herbivore interactions in multiple biological organizations.

Filling the gap 2: identifying key processes of blue carbon

Climate change has resulted from increased concentrations of carbon dioxide (CO_2) in the atmosphere, from $\sim$ 277 ppm in 1750, the beginning of the Industrial Era, to 393 in 2012. This will very likely cause future catastrophes (IPCC 2014a). Atmospheric carbon is removed by photosynthesis and stored in vegetation and soil within terrestrial ecosystems, which is so-called "green carbon". Vegetated coastal ecosystems such as mangroves, seagrass meadows, and salt marshes have long been recognized as some of the most productive natural ecosystems. They are considered blue carbon ecosystems, and long-term rates of carbon sequestration in these systems are much greater than green carbon (Mcleod et al. 2011). Therefore, they are thought to provide climate change mitigation benefits in view of their ability to store carbon (C) in their living biomass and in sediments (Nellemann et al. 2009; Tokoro et al. 2014; Miyajima et al. 2017). There is a need to correctly map and document vegetated habitats for their C storage to obtain a better and more reliable picture of the role of these ecosystems in climate change mitigation (Duarte et al. 2013).

C sinks in natural systems are primarily determined by processes through which plants uptake and use atmospheric CO_2 (Arrigo 2004). There is a need to understand how these vegetated habitats function as natural C sinks and how they are affected by underlying factors (Laffoley and Grimsditch 2009; Mcleod et al. 2011). Although accumulation rates of organic C in living biomass and sediments of blue C ecosystems have been estimated (Donato et al. 2011; Lavery et al. 2013; Shih et al. 2015), little is known about C sink mechanisms in natural systems (e.g., Lin et al. 2017). Very few studies have attempted to quantify and integrate relevant processes leading to C storage. To obtain a more accurate global storage rate estimate for blue C, reliable estimates are needed of these processes or C budgets across a broad geographic range and of the C storage potential of dominant plant species. Global analysis has demonstrated the accelerating annual loss of seagrasses from a median of 0.9% before 1940 to 7% since 1990, but there have been data deficiencies in the Asia–Pacific region (Waycott et al. 2009). This might undermine certainty in the global estimate of blue carbon emission

(Pendleton et al. 2012), owing to habitat destruction of coastal ecosystems. This region has the greatest species richness and the largest area of mangroves and seagrasses in the world (Ricklefs and Latham 1993). Many species grow together in mixed forests or meadows. A multispecies vegetated habitat may exhibit a higher production rate than a monospecific habitat (e.g., Erftemeijer and Stapel 1999). Empirical work on processes in multispecies vegetated habitats is needed to improve understanding of the contribution of vegetated coastal ecosystems to blue C. However, there are numerous gaps in the available literature for the Asia–Pacific region. This region is densely populated and home to $\sim$ 4.3 billion people (60% of the global population). Human activities have greatly modified environments between terrestrial and coastal zones (Regnier et al. 2013). Understanding the impacts of human activities on vegetated habitats that produce blue C will improve C cycle estimation. However, studies of factors controlling C budgets and identification of key processes are still lacking.

A recent study on the seagrass C budget at Dongsha Island in the South China Sea demonstrated that leaf production of multispecies seagrass meadows was greater than that of monospecific beds (Huang et al. 2015). Surprisingly, sediment organic C storage was much less than the global median value (Fourqurean et al. 2012). The lower sediment organic C stock was likely attributable to a higher detrital decomposition rate, because at least 60% of the leaf and belowground detritus was decomposed (Fig. 3). Similarly, a large proportion of mangrove ground litter was decomposed, but only a small proportion was buried in soil (Bouillon et al. 2008). Detrital export was also a key process in their C budgets, whereas the contribution of herbivory was relatively minor (Chiu et al. 2013). Despite this, there are still few field measurements of decomposition, export, or herbivory. Reliable estimates are particularly needed in the Asia–Pacific region to estimate the organic C storage rate. Quantifying the C budgets is important to better understand the global C cycle, support the development of climate policies, and project future climate change scenarios (Le Quéré et al. 2013). We suggest that the quantification of C budgets should consider habitat variability and processes across all seasons to ensure unbiased estimation. An understanding of the C budget over time and interannual variations associated with El Niño/La Niña (El Niño–Southern Oscillation) are necessary to understand and quantify climate-C feedbacks.

Filling the gap 3: adaptation to global biodiversity redistribution

Anthropogenic climate change has been driving species redistribution at an unprecedented scale and rate since the Last Glacial Maximum (IPCC 2014b). This will reorganize communities, altering ecosystem functions
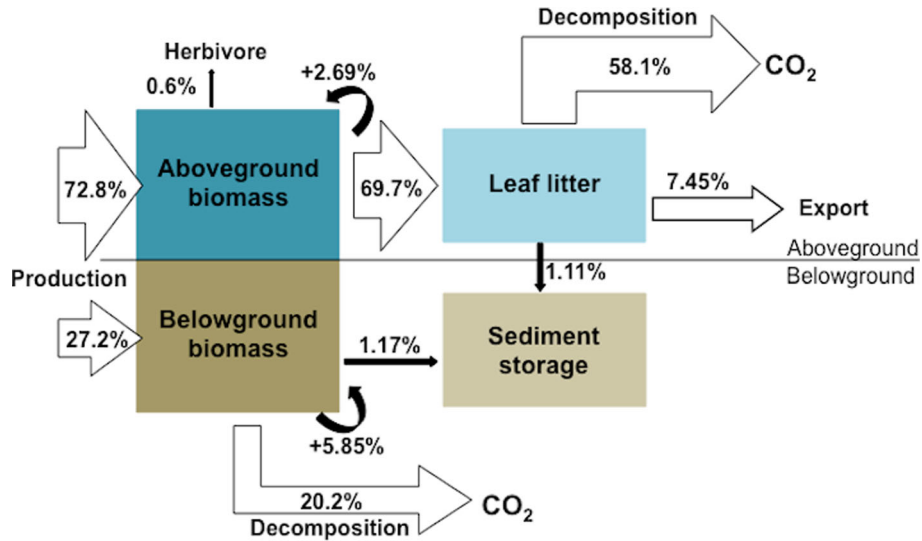
Fig. 3 Integrated annual budget of seagrass carbon fluxes in lagoon of Donogsha Island (Gap 2). Percentage values represent fraction of total primary production (1482 g C m^{-2} year^{-1}). Differences in box size do not imply differences in standing C stock. Figure is recalculated from Huang et al. (2015)

and services, resulting in climate feedbacks (Pecl et al. 2017). It is clear that responses of ecological systems would have tremendous impacts on socioecological systems, such as increasing risks of species extinction (Rohr and Raffel 2010; Urban 2015), infectious disease (Wu et al. 2016), or food insecurity (Bell et al. 2016). Research into biodiversity redistribution has grown rapidly in the past two decades and has emerged as a new field of multidisciplinary study (Bonebrake et al. 2017). Focusing on climate-driven species redistribution, we list herein several knowledge gaps of biological responses under climate change, required shifts of conservation practices, and links between ecological and social systems in need of strengthening.

Biodiversity redistribution is evident in wide taxonomic groups, with terrestrial species estimated to move poleward by 17 km and upward by 11 meters per decade (Chen et al. 2011), and marine species by 72 km per decade (Poloczanska et al. 2013). The global redistribution of species, however, shows great variability and idiosyncratic responses, depending on species' physiological tolerances, dispersal constraints, and interactions with other species. Moreover, species redistribution can be accelerated by short-term human activities such as harvesting (e.g., Hsieh et al. 2008; Kuo et al. 2016). However, there are huge gaps between observation and mechanistic understanding (Fig. 4). Studies of idiosyncratic range shifts largely focused on intrinsic species traits (Angert et al. 2011; MacLean and Beissinger 2017) and have not yet provided conceptual or theoretical frameworks to fully consider factors involved in the process of range shifts (Estrada et al. 2016). Some aspects at species and community levels may at least be understood in biodiversity redistribution ecology, as discussed below.

Evolutionary dynamics of range shifts

Thermal performance is key to understanding species response and interaction under climate change (Dell et al. 2011). Intraspecific variation of thermal traits may imply the capacity for range shifts. However, the pattern and genetic basis remain to be explored. Measuring genetic structure together with range dynamics provides a
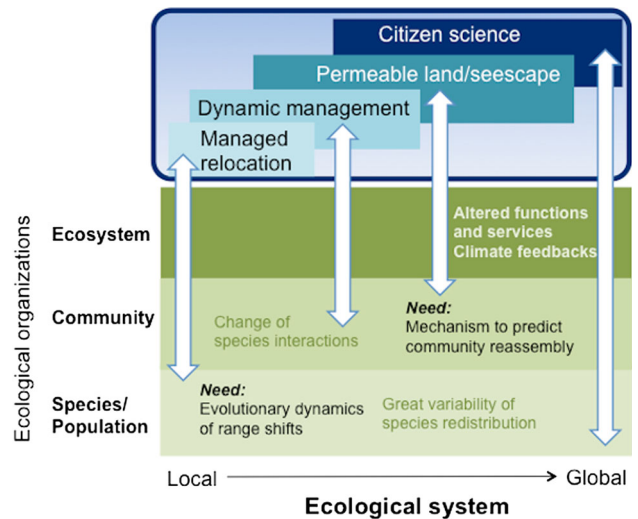
Fig. 4 Key effects of global biodiversity redistribution and links to conservation practices (Gap 3). Climate change has resulted in range shifts of many taxonomic groups across the globe. The effect of species redistribution varies with ecological levels and spatial scales. We show some of the knowledge gaps in the framework. We need new paradigms of conservation practices, which strengthen links between ecological and social systems at different levels, to prepare for an overwhelmingly changed earth

wealth of information on evolutionary adaptation (Sexton et al. 2011).

Mechanism of community reassembly

Predicting future species composition remains a major challenge for forecasting biodiversity redistribution. Competitive, trophic, or mutualistic interactions may impede or accelerate species range shifts (Urban et al. 2012; Blois et al. 2013; Alexander et al. 2015). Traditionally, studies of species distribution at large scale and species interaction in local communities use different techniques, theories and datasets (Staniczenko et al. 2017). The lack of predictability for biodiversity redistribution highlights the need to bridge the divide and deepen our mechanistic understanding of community reassembly. Several lines of work have improved dynamic macroecological modeling. A semi-mechanistic approach incorporates key processes of dispersal and community assembly to predict present and future species composition (Mokany and Ferrier 2011; Mokany et al. 2012). Another approach combines Bayesian networks and distribution modeling, which explicitly captures effects of biotic interactions as conditional dependencies among species presence (Staniczenko et al. 2017).

New paradigm of conservation practices

Conventional biodiversity conservation relies predominately on static protected area (PA) systems, targeting specific species or ecosystems. Given that species redistribution will continue in the foreseeable future, conservation strategies must accommodate changing biological communities.

Permeable landscape and seascape

To improve the likelihood of successful dispersal, a PA system should be designed for in situ conservation and consider the surrounding environment to allow permeable landscapes and seascapes. PA systems can be made more climate-resilient by including heterogeneities of microclimate and microhabitat, or areas with slow climate change where climate refugia are more likely to occur (Burrows et al. 2014). In fact, current PA systems may have had important roles in facilitating range expansion and maintaining populations (Thomas et al. 2012; Johnston et al. 2013). However, the failure of PA systems is likely (Scriven et al. 2015). Improving overall connectivity is crucial for long-term persistence of biodiversity. Partnership with private sectors, including land owners and local communities, should be mainstreamed in conservation practices.

Managed relocation

Warming-induced biodiversity redistribution reshuffles biological communities and accelerates the risk of extinction (Urban 2015). Conservation strategy should move toward "managing change" rather than trying to maintain past known communities, given the reality of rapid environmental and biological changes and the goal of prioritizing the preservation of global biodiversity (Thomas 2011). Relocating species to climate-suitable areas outside their native ranges is an option to prevent species extinction. However, this option is often ignored, partly because of the fear of unpredictable consequences of "invasive" species in receiving communities (Ricciardi and Simberloff 2009). In practice, we can ask whether the risk is truly unpredictable and what type of species should be considered. The most severe consequence of new biological interaction may be species extinction. In particular, this involves translocating species to a different biogeographic region or very isolated environment, such as oceanic islands. These are not cases for managed relocation, usually because climate-suitable areas in the region are considered. Moreover, translocation should give priority to highly vulnerable species with limited chance to sustain in situ. Essentially, threatened species that could thrive elsewhere but are currently hindered by weak mobility or inhospitable surroundings may be suitable targets (Thomas 2011).

Strengthen links between ecological and social systems

The pervasive effects of species on the move transcend systems between natural systems and human societies. However, the necessary links and communication between social and ecological systems are generally weak, making conservation practice inefficient (Heller and Zavaleta 2009). Understanding the pattern and process of biodiversity redistribution not only prevents species extinction but strengthens resource management.

Changing distributions of pollinators, pests, or pathogens pose great challenges to agriculture, fisheries, and aquaculture (Bell et al. 2016). New paradigms of dynamic management and monitoring schemes will be critical, such as the following: (1) dynamic ocean management (DOM). Management responses adapt to the shifting nature of the ocean and its users. Implementation requires technological and policy capacities and enables wise use and proactive management (Hobday et al. 2014; Dunn et al. 2016). (2) Early warning systems supported by citizen science. The collection of spatially and temporally explicit data of species and environmental change facilitates the aforementioned research and modeling, which supports science-based decision-making. Citizen science projects led by the Cornell Lab of Ornithology (http://www.birds.cornell.edu/page.aspx?pid=1664) and Range Extension Database and Mapping Project (http://www.redmap.org.au/) led by

the Institute for Marine and Antarctic Studies are good examples of citizen involvement (see also "Filling the gap 4: importance of ecological theory and biological monitoring"). These projects focus on birds and marine taxa, respectively, detecting species on the move and community reassembly at large spatial and temporal scales.

The cross-cutting nature of biodiversity redistribution requires collaboration among experts in ecology and climate, data, and the social sciences. Monitoring schemes bridge science, private and governmental sectors, further improving the understanding and management of socioecological systems. Adapting to altered socioecological systems requires governance that prepare for overwhelmingly volatile conditions.

Filling the gap 4: importance of ecological theory and biological monitoring in community organization

Mechanisms determining species assembly and resultant community structure are long-standing subjects in ecology. To understand community organization in diverse systems, therefore, a number of ecological theories have been proposed. These theories are roughly categorized into deterministic and stochastic views.

The deterministic view focuses on environments as direct determinants of community structure, because these can directly affect physiological processes of species or, indirectly, biological interactions such as competition and predation (e.g., Weng et al. 2017). Some community ecology theories, e.g., "Paradox of plankton" (Hutchinson 1961), theories of keystone species (Paine 1969), and trophic cascade (Carpenter and Kitchell 1996), rely on such deterministic forces. Even the alternative stable state theory (Scheffer et al. 1993) relies on a deterministic view, because the stability landscape of equilibria changes with environmental conditions. Ultimately, this view implies that abiotic environmental factors and biological interactions act as filters that shift or select organisms as members of a local community. In other words, these filters characterize niche space in a local community, and so the deterministic view implicitly assumes that the community is structured by niche limitation.

The stochastic view focuses on dispersal abilities or probabilities of immigration success of species more than on environments and biological interactions, and address the unequal ability and probability of organisms to access a local habitat (MacArthur and Wilson 1967). Diversity of life histories, e.g., small-size vs. large-size propagules and planktonic propagules drifting over long periods vs. early settlers, indicates that dispersal abilities and therefore probabilities of immigration success to new habitats vary between organisms (Lockwood et al. 2005). It is also likely that the immigration success of an organism depends on the meta-population structure, i.e., distance and size of a source population. If these factors are dominant, the local community would be shaped by

the dispersal probabilities of organisms. The theory of island biogeography (MacArthur and Wilson 1967), lottery competition (Chesson and Warner 1981), and neutral theory (Hubbell 2001) are in line with the stochastic view. Implicitly, that view implies that the community is structured by dispersal limitation.

In community assembly, however, deterministic and stochastic views are probably not exclusive to each other. Rather, both niche and dispersal limitations operate subsequently to shape community structure. To be a member of a local community, each species must pass through a number of filters, some of which are deterministic but others stochastic (Fig. 5). However, our understanding of the relative importance of deterministic (or niche limitation) and stochastic (or dispersal limitation) forces is still limited, probably because of few opportunities to experimentally examine meta-communities at large ecosystem scale.

One such opportunity is a powerful natural disturbance that has ecological impacts at regional or multi-ecosystem scales. For example, the Great East Japan Earthquake in 2011 and resulting large tsunamis struck the Pacific coast of eastern Japan and affected coastal ecosystems at regional scale (Shimada 2016). In some tidal communities, a number of benthic animal species dramatically decreased immediately after the tsunamis (Urabe et al. 2013). In tidal flats where environmental conditions did not change substantially, after an initial increase of species richness over 2 years, species richness and composition returned to levels prior to the tsunamis in the fourth year. These findings imply that niche limitation was important in structuring communities in the later stage (by 2014) through environmental filtering and/or biological interactions. In fact, in 4 years, species compositions roughly returned to those previously found in certain tidal flats before the tsunami (Nishita et al. 2016). In contrast, in tidal flats where environmental conditions changed greatly because of the tsunamis, communities also changed and did not show a recovery to the original state for at least 4 years after the tsunamis (Kanaya et al. 2012, 2015). These results sug-
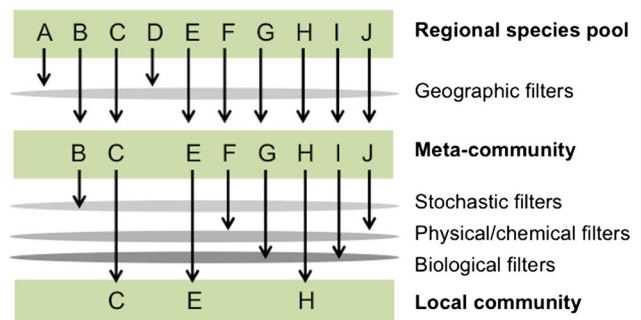
Fig. 5 Diagram of the community organization (Gap 4). Species in a local community are selected through various deterministic filters within local environments and stochastic filters between local communities and meta-communities. Species in meta-communities are determined by species in regional pool that pass through geographic filters

gest that the tidal communities are assembled and shaped mainly by niche limitation. However, in any tidal flat, community structures examined in these studies never reached the original ones before the tsunamis. Thus, it is premature to conclude that niche limitation plays a more pivotal role than dispersal limitation (Nishita et al. 2016). Clearly, we need more long-term observation of the communities.

Climate models have forecast that owing to putative climate change, there will be more frequent large or extreme natural disturbances such as typhoons, floods, and drought (e.g., Easterling et al. 2000). Ironically, such natural disturbances may provide an opportunity to study how communities are assembled and shaped. To take full advantage of this opportunity, we must develop ecological theories considering both deterministic and stochastic views that can be feasibly used to analyze empirical data (e.g., Chang et al. 2017). More importantly, we need community data before the disturbance. Otherwise, we can do nothing, even if we have sophisticated ecological theories at our disposal.

One of the promising approaches to long-term observation programs would be related to citizen science (Urabe et al. 2013; Kobori et al. 2016; Nishita et al. 2016). After the tsunamis, we implemented a monitoring program for benthic communities at Sendai Bay every year, with over 500 citizen volunteers over 5 years (Ito 2017). The program provided benefits to citizens (e.g., playing with mud and animals (refreshment), learning the ecology and ecosystems of animals, the environment, and biodiversity literacy, and conservation consensus) and ecologists (e.g., outreach of ecological importance, sharing with society, feedback to our science, reconsideration of conservation needs, and donations from private sectors) (Ito 2017). It therefore functioned to bridge a gap between citizens and ecologists and solved a human-power shortage in obtaining high-quality large datasets (Fig. 1). These data will certainly be treasures for future ecologists to understand community organization under climate change, and provide a basis for science-based ecosystem management.

Social Security Research (IPSS) 2014; MLIT Japan 2015). Also, existing infrastructures, including those for disaster prevention, have been aging, and the predicted expenditure for renewal of those infrastructures in the next several decades continues to increase (MLIT 2012).

Recently, increasing attention has been paid to ecosystem-based adaptation to risks of climate change, with a focus on lower economic cost and multiple functions of ecosystem services (e.g., CBD 2015; UN Office for Disaster Risk Reduction 2015; MOE Japan 2016). However, in local communities, this adaptation can sometimes be rediscovered in the traditional wisdom of local people (e.g., Fukamachi et al. 2009; MOE Japan 2016). Ecosystem-based disaster risk reduction (Eco-DRR) takes advantage of both multiple ecosystem services and disaster risk reduction by, for example, reducing the exposure of human systems through land-use management (SCJ 2014; The Royal Society 2014) (Fig. 6). Population decline provides ample opportunity for implementing Eco-DRR in Japan. However, quantitative analysis of the multi-functionality of Eco-DRR is still in its infancy (SCJ 2014; The Royal Society 2014), so detailed technical comparison with conventional hard-engineering countermeasures is not yet feasible. This hampers the social implementation of Eco-DRR.

To understand how the ecosystem-based approach can simultaneously reduce disaster risks and restore/conserve biodiversity, we have been studying how the interaction of natural disturbances and land use influences disaster risks and biodiversity in the Lake Mikata watershed of Fukui Prefecture. This area has flooding every 5 years on average, and some new countermeasures are being planned. Our preliminary results suggest that Eco-DRR can produce win–win situation for biodiversity conservation and disaster risk reduction. Our species distribution modeling with both land use and flood risk (i.e., flood disturbance of organisms) as explanatory variables showed that some organisms should benefit from the land-use pattern, such that their habitat is selectively conserved in the flood-
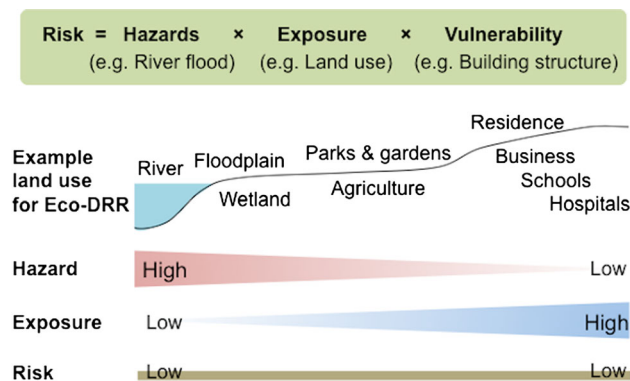
Filling the gap 5: adaptation to natural disasters: how does the interaction of disturbance and land use influence biodiversity and disaster risks?

In addition to global-scale impacts, future climate change is also predicted for Japan, including warming and increased heavy rain (MOE Japan 2014). The impacts of climate change extend to various natural and human systems worldwide (IPCC 2014b). Among these, natural disasters have been increasing, partly because of contemporary climate change (IPCC 2014b; Munich Re 2015). Adaptation to disaster risk is important for the sustainability of both local and global human societies. Coincident with climate change, many local communities in Japan are experiencing shrinking populations, leading to the abandonment of farmlands, houses, and other intensive land use (National Institute of Population and

Fig. 6 Risk of natural disasters and Eco-DRR (Gap 5). Risk of natural disasters consists of hazards, exposure, and vulnerability. Ecosystem-based disaster risk reduction (Eco-DRR) takes advantage of both multiple ecosystem services and disaster risk reduction by managing land use of towns and cities, as exemplified here

prone area, thereby avoiding the flood risk to human society. We are currently examining what the Eco-DRR potentially provides in terms of multiple ecosystem services, including provisioning, regulating, and cultural services. By integrating their results, we wish to propose how the Eco-DRR can be compared to the conventional hard-engineering countermeasures in terms of reducing the risk of natural disasters and multi-functionality of ecosystem services.

What are the benefits of filling the gaps?

Here, we summarize how filling the gaps in socioecological systems is beneficial for ecological science and society (e.g., citizens and governments). The examples above illustrate two different types of gaps in our understanding of socioecological systems (Fig. 1). Ecology has developed through repeated cycles of diversification into sub-fields and their reintegration. In the last two decades, the integration of subdivided fields in ecology has facilitated great progress in ecology. Actually, such approaches link different levels of ecological organizations, from community to ecosystem (e.g., Loreau et al. 2001; Ehrenfeld et al. 2005; Gap 2) and evolution to community (e.g., Matthews et al. 2014; Gap 1). Eco-evolutionary approaches will also increase understanding of complex social behaviors (Harrington and Sanchez 2014). An urgent need is to scale up climate-change biology at organism and species levels to community and ecosystem levels (Gap 3).

Filling the gaps between ecological organization and processes is crucial for ecologists to provide citizens with an overall picture of ecological sciences. Ecologists cannot convince citizens and other scientists that ecology is often more valuable than the natural history of animals, plants, and microbes without sharing the full picture of the ecological sciences. Highlighting links and feedbacks between multiple ecological processes enables citizens to better understand that ecological conservation of ecological processes (e.g., the potential of evolution and pollination service) and ecosystem processes (e.g., primary production and nutrient cycling) within a community in a certain local or regional area, and global biodiversity conservation, are much more important than separately protecting individuals of endangered species. In other words, filling the gaps between different ecological organizations not only advances ecological science but also improves communication between ecologists and citizens. Similarly, citizen science is not a unidirectional information flow from ecologists to citizens; data obtained by citizens will furnish precious materials for cutting-edge research to advance ecological theories (Gap 4). Covering multiple spatial scales in each ecological organization is also a promising approach to advance ecology (Gap 2 and Gap 4) and enhance the connection between ecological and social systems (Gap 5). Filling the gaps between multiple spatial scales will make it easier and more effective for local and regional governments (from city and county to nation) to design effective ecosystem management strategies dependent on spatial scale.

Next challenges

We now have an updated map that better highlights interrelationships among components within the socioecological system (Fig. 1), but this map remains incomplete. The following are a few examples of further steps to improve our map. Gaps between the proposed gaps should be also filled. For example, eco-evo feedback concepts and phenotypic plasticity in species traits (Gap 1) should be incorporated into the framework of the global redistribution of biodiversity (Gap 3). In addition to proposing brand new ideas and accumulating more empirical data, what we need to advance ecology includes sharing of existing theories and concepts from different scientific fields. This often boosts our understanding [e.g., unified theory of biodiversity (Hubbell 2001), originally from the neutral theory of molecular evolution (Kimura 1968); evolutionary game theory (Smith 1974) that was developed from game theory in mathematics and economics (Nash 1950)]. We need positive attitudes to fill the gaps between scientists in diverse fields of academics (e.g., Liu et al. 2017). Sharing data is a growing trend in ecological sciences and other fields. The data paper section has been published in *Ecology*, *BMC Ecology*, *PLOS ONE*, and *Ecological Research* (e.g., Iwayama et al. 2017). There are many open ecological databases [FishBase http://www.fishbase.org/search.php; GPDD (Inchausti and Halley 2001); TRY (Kattge et al. 2011)] and genomic information [e.g., KEGG (Kanehisa and Goto 2000), MBGD (Uchiyama et al. 2010), and SILBA (Quast et al. 2013)]. The International Long-Term Ecological Research East Asia–Pacific Regional network has accumulated valuable datasets (Kim et al. unpublished). The next step may include developing a framework to share research ideas and hypotheses before completion of research projects (e.g., Royal Society Open Science). Such new frameworks for sharing ideas will accelerate the filling of gaps between ecologists. Eventually, via gap-filling, the application of new ecological sciences arising from these continuous challenges to local and regional societies in East Asia will improve environmental management at various government scales under human-driven environmental changes (e.g., Laplace et al. 2017; Lin et al. 2017).

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