THE IMPORTANCE OF COLLABORATIVE LEARNING AND RESEARCH AMONG CONSERVATIONISTS FROM DIFFERENT OCEANIC ISLANDS

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RÉSUMÉ.— *L'importance de la collaboration entre conservationnistes des différentes îles océaniques pour l'enseignement et la recherche sur la biodiversité.*— Dans cet essai, je plaide pour que les conservationnistes de la biodiversité dans les îles s'engagent plus régulièrement dans un processus d'enseignement inter-îles et tirent profit des opportunités de recherches apportées par des îles confrontées à des problèmes similaires de conservation. En particulier, dans un monde évoluant rapidement, les enseignements à partir de nombreux sites sont importants pour s'assurer que les conservationnistes prennent bien en compte des changements imprévus et l'évolution vers de nouveaux écosystèmes. Les occasions d'interactions entre conservationnistes de différentes îles sont croissantes, mais les échanges de connaissances restent souvent informels et faibles. Une des raisons tient peut-être dans les difficultés de généralisation des connaissances à propos de systèmes qui sont souvent très particuliers. Il n'est de ce fait pas facile d'extrapoler des résultats obtenus sur un site à un autre site. Cependant, les médecins par exemple sont aussi confrontés à ce problème de construction d'un corpus de connaissances pouvant être transférable d'un patient au suivant malgré l'unicité de chaque patient. Comme les médecins, les conservationnistes doivent apprendre comment partager un ensemble de connaissances et d'expertises relatives à des cas particuliers mais tout de même transférables entre les modes de gestion. Quatre exemples de sujets illustrant les enjeux de conservation et nécessitant une approche systématique inter-îles sont présentés: (1) la prévision des risques d'invasions par les plantes, (2) les impacts des rats introduits sur la flore et la faune indigènes, (3) les interactions mutualistes entre plantes et animaux, (4) la restauration des habitats. Je termine l'article en insistant sur le fait que le succès de la recherche et de l'enseignement inter-îles dépend de collaborations continues sur le long terme. Une biodiversité insulaire unique est en train de disparaître rapidement. Les conservationnistes insulaires sont confrontés à la tâche difficile d'inventer et de mettre en œuvre de manière simultanée et sur une période très courte de nouvelles stratégies concernant de nombreuses menaces en interaction et évolution rapide, qui affectent des milliers d'espèces insulaires menacées. Pour sauver cette biodiversité insulaire de l'extinction, un effort global d'échanges de connaissances et d'actions entreprises doit être réalisé de manière urgente.

SUMMARY.— I argue in this essay that conservationists on islands should engage more regularly in systematic cross-island learning and capitalise on the research opportunity provided by replicated islands around the world that are faced with similar nature conservation problems. In a rapidly changing anthropogenic world in particular learning across multiple sites is important to ensure that conservationists are not blind to unexpected future changes and novel ecological patterns and processes. There are increasingly opportunities for conservationists from different islands to interact but knowledge exchange remains often informal or piecemeal. One reason why cross-island learning is relatively rare may be the difficulty to generalise knowledge about real-world systems that are often highly idiosyncratic. It is thus often not clear how the insights gained in one place can be applied in another. However, medical doctors for instance are also faced with the challenge of building up a body of knowledge that is transferable from one patient to the next despite the uniqueness of each patient. Like doctors conservationists need to better learn how to form a shared pool of knowledge and expertise that is context-sensitive but still transferable between management cases. To indicate the range of conservation issues on islands that await a more systematic cross-island learning approach I discuss four examples: (i) predicting plant invasion risks, (ii) impacts of alien rats on native flora and fauna, (iii) mutualistic plant-animal interactions, and (iv) habitat restoration. I end the article by emphasising that successful cross-island research and learning depends on long-term continuous collaborations. Unique island biodiversity is rapidly disappearing. Island conservationists are confronted with the taunting task of devising and implementing new conservation strategies that address at once and in a very

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short time span many different interacting and rapidly changing threat factors, which affect thousands of threatened island species. To save much island biodiversity from extinction a global and concerted learning and action effort is urgently needed.

Oceanic islands have served research in biogeography, ecology, evolutionary biology and conservation as model system since the early days of these research fields (Darwin, 1859; Whittaker & Fernández-Palacios, 2007). Islands were for instance crucial for the formulation of the theory of island biogeography (McArthur & Wilson, 1967) or the observation of evolution in action (Grant, 1998). However, while in particular biogeographers regularly use the opportunity of many replicated oceanic islands for comparative research, conservation research has to date mostly focused on single islands or archipelagos (Caujapé-Castells *et al.*, 2010; Kueffer & Fernández-Palacios, 2010).

I argue in this essay that conservationists on islands should engage more regularly in systematic cross-island learning and capitalise on the research opportunity provided by replicated islands around the world that are faced with similar conservation challenges. Sometimes studies involving different islands within an archipelago (e.g. Chiarucci *et al.*, 2010) or different archipelagos of a particular oceanic region such as the Pacific, Caribbean Sea or Western Indian Ocean (Kueffer *et al.*, 2004; Mueller-Dombois & Daehler, 2005; Maunder *et al.*, 2008) can be productive, but often far apart islands share similar ecologies and conservation challenges – e.g. high elevation islands such as Tenerife in the Atlantic and Hawaii in the Pacific situated at opposite ends of the planet (e.g. Domínguez Lozano *et al.*, 2010) – and collaborations are therefore needed at a global scale (Caujapé-Castells *et al.*, 2010; Kueffer *et al.*, 2010a).

There are increasingly opportunities for island conservationists to meet at workshops, exchange information via the Internet, or raise funds for projects that involve research partners from different islands. However, knowledge exchange remains often informal or piecemeal. A reason for the lack of collaborative learning may be that an awareness of what cross-island learning could mean is missing. I argue in a first paragraph of this article that it is useful to think of cross-island learning in analogy to the formation of a shared expertise among medical doctors. Doctors are also confronted with the challenge that each patient is an individual case but they nevertheless must try to find commonalities among individual cases as a basis for medical progress. In a second part I then provide an outlook on some exemplary conservation issues on islands that await such a more systematic cross-island learning approach.

GENERALISATION OF KNOWLEDGE IN CONSERVATION BIOLOGY

Conservationists are in their work confronted with the challenge of understanding and managing real-world problems for which the specific factors that characterise a particular system matter a lot, i.e. conservation problems are often highly idiosyncratic. It is thus often not clear how the insights gained in one place can be applied in another. Is a conservation strategy that saved a critically endangered plant on one island also effective for a different species on another island? Does the same habitat restoration approach work for two different sites? The same difficulty of generalising knowledge affects also other sciences besides conservation biology such as ecology, environmental sciences, medicine or social sciences and I use the term "real-world sciences" or "real-world problems" to denote this type of research. These sciences deal with study systems in the real-world that are characterised by myriads of known and unknown interacting factors and are difficult to manipulate and replicate in contrast to experimental systems in laboratory research.

Conservationists tend to fall into two groups that use different strategies to circumvent the difficulties of generalising knowledge from multiple real-world cases. The first group attempts to identify universal principles that equally apply to a broad range of species, ecosystems or

management problems despite real-world heterogeneities. Examples of this 'one answer fits all problems' thinking are the use of species-area relationships for predicting extinction risks ("a certain reduction of habitat area will lead to a predictable species loss independent of habitat or species") (e.g. He & Hubbell, 2011), or the enemy release hypothesis in invasion biology ("invasive alien species generally profit in a new area from the release from natural enemies such as herbivores or soil pathogens that are not present in the invaded range") (e.g. Chun *et al.*, 2010). The second group of conservationists attempts to develop a context-specific understanding of each species, ecosystem or management case separately through in-depth case studies (Orians *et al.*, 1986; Shrader-Frechette & McCoy, 1993; Kueffer, 2006; Plowright *et al.*, 2008). Kueffer *et al.* (2010b) for instance devised a management strategy for a 'novel' forest ecosystem dominated by alien *Cinnamomum verum* trees in Seychelles by taking into account context-specific factors related to the habitat (e.g. phosphorus-poor soils, Kueffer *et al.*, 2008; Kueffer, 2010a), the disturbance history of the site (e.g. highly fragmented native plant distributions, very high abundance of *C. verum* on a landscape scale, Kueffer & Vos, 2004), the biology of the dominant invading species, *C. verum* (e.g. strong belowground root competition, nutritious fruits, Kueffer *et al.*, 2007; Kueffer *et al.*, 2009), and characteristics of the native flora (e.g. an advantage of regenerating native compared to invasive juvenile plants when both above- and belowground resources are in short supply (Schumacher *et al.*, 2008; Schumacher *et al.*, 2009).

While both strategies are needed and have merits, they also both have important limitations. The first approach – 'one answer fits all problems' – suffers from the problem that in conservation general laws are rarely applicable to concrete management cases without accounting for the particular context. In invasion biology for instance meta-analyses indicate that enemy release does sometimes but not generally help to explain the rapid spread and high competitiveness of alien species in a new area (e.g. Chun *et al.*, 2010). The second approach – 'every problem requires a different answer' – suffers from a lack of predictive power and ability to learn from research and management in other places. In particular in a rapidly changing anthropogenic world learning across multiple case studies is important to ensure that conservationists are not blind to unexpected future changes and novel ecological patterns and processes (e.g. Kueffer, 2010b; Dawson *et al.*, 2011; Estes *et al.*, 2011).

The middle ground between general laws and in-depth case studies is surprisingly poorly explored in conservation biology, but such knowledge at intermediate generality is particularly relevant for collaborative learning and research among conservationists. It is useful to think of such learning across case studies in analogy to the formation of a shared expertise among medical doctors. Each patient is an individual case, and to help a patient, doctors have to integrate their general medical knowledge and their in-depth experiences from many previous patients with similar but not identical health problems. A doctor will neither think that each patient is unique nor expect that the same treatment equally applies to every patient. What makes a medical doctor a reliable expert under these conditions, and therefore is a requirement for the diploma of a doctor, is his extensive experience from treating thousands of different patients. In nature conservation it is for practical reasons not possible to gain such extensive experience from multiple sites to the same degree, but the need for knowledge that is condensed from diverse real-world experiences is the same as in medicine.

There is a growing body of research in fields such as artificial intelligence (e.g. Beierle & Kern-Isberner, 2006) or philosophy (e.g. McKay Illari *et al.*, 2011) that is interested in a formal understanding of the causal reasoning that is characteristic of experts working on real-world problems. This literature cannot be reviewed in this article, but I will hint at some central ideas. The key challenge for scientific understanding in real-world sciences is that a cause can have different effects under different conditions. For instance, a cause may exert an effect only in combination with other factors or under particular auxiliary conditions, or the presence of particular factors can destroy or overwhelm the operation of a cause. It is thus in real-world sciences necessary to understand how the operation of a cause changes when other factors of the real-world interfere. In other words, understanding a cause in the real-world requires a theory that explains how the cause operates in different contexts.

Such context-sensitive theories can be formalized in different forms, and I mention in the following three types for illustration: (i) a comprehensive compilation of those explanations that work in some cases, (ii) a hierarchical multi-tier framework, or (iii) a rule-based expert system. The simplest form of cross-case knowledge is a comprehensive compilation of those explanations that work in some cases possibly including a qualitative understanding of the context-dependence of these explanations: for instance a synthesis of threat factors affecting native plants on oceanic islands (Caujapé-Castells *et al.*, 2010). Such a synthesis can be used like a checklist when addressing a new management problem but it does not give much guidance on how to proceed in evaluating different potential causes. A more sophisticated structuring is to arrange explanatory variables in a hierarchical multi-tier framework (e.g. Ostrom, 2007). The higher levels ('tiers') of such a framework are considered more generalizable (i.e. less affected by different contexts), while lower levels are consecutive specifications of the higher-level factors. The user of such a conceptual framework starts with the first-level factors and tests how far he gets in explaining a particular case. Depending on the outcome, he may then need to move on to lower-level factors and add them to the explanation until he gains a satisfactory understanding of a particular case. The general theory of biotic invasions can be understood as such a hierarchical, multi-tier conceptual framework that is built on four first-tier variables: invasiveness (the traits of the invasive species), invasibility (the characteristics of the invaded habitat), propagule pressure (processes related to the introduction of seeds or other propagules to a site) and residence time (the time since first introduction) (Lonsdale, 1999; Kueffer & Hirsch Hadorn, 2008; Richardson, 2011). For each of these four first-tier variables more specific factors have been defined on a second lower level. Invasiveness is further specified by a range of species traits such as growth rate, dispersal mechanism, environmental tolerance (e.g. shade tolerance), or impact traits (e.g. N_2-fixation or fire-related traits). Second-tier variables for invasibility include the role of abiotic conditions such as resource availabilities or climatic conditions, natural and anthropogenic disturbances, or biotic composition. For propagule pressure the timing, frequency, magnitude and quality of individual introduction events matter (Lockwood *et al.*, 2005). In the context of residence time the relevance of processes related to population dynamics or evolutionary adaptations have for instance been discussed. Foxcroft *et al.* (2011) have recently demonstrated how this hierarchical framework can indeed be successfully applied to a specific invasion problem: the invasion of *Opuntia stricta* in the Kruger National Park in South Africa. Still more formalized formats of cross-case knowledge are rule-based expert systems that are the subject of intensive research in the field of artificial intelligence (e.g. Beierle & Kern-Isberner, 2006). Such expert systems do not only order explanatory variables in a way that facilitates the search process for an explanation of a new case but include also rules on how to proceed in this search for an explanation; e.g. through a sequence of if-then statements. Decision-support systems, such as those developed for predicting the ecological risks of invasive species or genetically modified organisms (GMO) (e.g. Andow & Hilbeck, 2004; Daehler *et al.*, 2004), are examples of rule-based decision-support tools that enable the prediction of the outcome in a particular case based on knowledge generalised across many cases.

It is in this article not possible to develop a comprehensive understanding of generalising knowledge across real-world case studies in conservation biology. But I hope that the paragraph helps to convey the idea that it is indeed fruitful to engage in intensive collaborative learning across multiple real-world case studies on specific research and management problems such as those discussed in the next paragraph.

COLLABORATIVE LEARNING AMONG ISLAND CONSERVATIONISTS

In the previous paragraph I have argued that environmental scientists are beginning to realize that beyond the identification of universal laws and the accumulation of in-depth case studies new forms of learning and theory formation are needed that facilitate the transfer of insights gained from one in-depth real-world case study to another. Our ability for cross-case learning in environmental research, and more specifically in conservation biology, is still in its infancy but island conservationists are in the privileged situation of working on a system

that is replicated many times worldwide. In this paragraph I will provide an outlook on four exemplary conservation issues on islands that await a more systematic cross-island learning approach. The examples are not exhaustively treated but rather selected aspects are discussed for illustrative purpose.

PREDICTION OF PLANT INVASION RISKS ON OCEANIC ISLANDS

Kueffer *et al.* (2010a) discuss in their global-scale study that species traits, habitat factors (including land use and anthropogenic habitat disturbances), the history of species introductions (propagule pressure), and residence time co-shaped plant invasion patterns on oceanic islands. In particular they show that most species were invasive only in one to a few islands although they were typically introduced to many more islands. This indicates that generalising about traits of invasive species (invasiveness) without considering the invasion context will not lead to reliable predictions of future plant invasion risks on oceanic islands. Therefore comparative analyses of multiple in-depth case studies that address the interactions of invasive species with the socio-ecological characteristics of the invaded islands are needed. For such integrative comparisons it is promising to conceptually depict an invasion as a sequence of different phases (Theoharides & Dukes, 2007; Richardson, 2011). A successful invasive species is first introduced to a new area (introduction), then establishes a self-reproducing population and spreads (establishment and spread), before it becomes abundant and persists in a natural area (invasion). In each of these phases different species traits and external factors are relevant (Theoharides & Dukes, 2007; Richardson, 2011).

Processes during the first phase – the introduction phase – are shaped by human behaviour. Humans transport alien species intentionally or unintentially over large distances to new areas (Hulme *et al.*, 2008). The positive correlation between the number of introduced and invasive alien species per island and socioeconomic factors such as the gross domestic product (GDP) is likely at least partly explained by higher introduction rates of alien species to islands with a higher population density or economic growth (Denslow *et al.*, 2009; Kueffer *et al.*, 2010a). In fact, the similarity of alien (but not necessarily invasive) floras among oceanic islands is generally high despite large geographic distances between islands (Kueffer *et al.*, 2010a), which is likely the result of similar past socioeconomic activities on different islands. Improving the prediction of invasion risks requires thus at first an understanding of the future socioeconomic development and trade relationships of island societies.

The second phase – establishment and spread – selects for invasive species that successfully establish from small population sizes, are effectively dispersed, and are ecologically plastic. Indeed many major invaders on islands have a high phenotypic plasticity compared to native species (e.g. Schumacher *et al.*, 2008, 2009) and have been able to spread from small founder populations with little genetic diversity – e.g. *Clidemia hirta* (DeWalt & Hamrick, 2004), *Miconia calvescens* (Le Roux *et al.*, 2008), *Pennisetum setaceum* (Le Roux *et al.*, 2007), or *Senecio madagascariensis* (Le Roux *et al.*, 2010). Ecological plasticity may be particularly relevant for invaders on islands where ecological conditions at introduction sites in the anthropogenic lowlands differ often strongly from those of invaded sites in the interior of islands (e.g. Haider *et al.*, 2010; Jakobs *et al.*, 2010; Alexander *et al.*, 2011). For improving generalisation about species traits conferring invasiveness in the establishment and spread phase it is thus particularly important to separately analyse species that depend on different dispersal pathways. For instance, those species that are introduced at a site that is ecologically and geographically distant from the site of invasion depend on high ecological plasticity and efficient dispersal, unintentionally introduced species often depend on an ability to establish and spread from small founder populations, while other species became invasive thanks to large-scale deliberate planting in the areas where they became invasive (e.g. in Hawaii, Woodcock, 2003). These invasive species do not depend on an efficient dispersal mechanism.

The fourth phase – invasion – is a result of tight interactions between species traits and habitat characteristics. Considering habitat context is therefore in this invasion phase especially relevant for improving prediction. A particular invasive plant species invades only some habitats but not others, and invaders of different habitats tend to have different traits (e.g. Llo-

ret *et al.*, 2005; Kueffer & Daehler, 2009). For instance, invaders of shaded environments such as a forest understory differ from invaders of open habitat (Martin *et al.,* 2009; Schumacher *et al.,* 2009). During the actual invasion phase a species will possibly have negative impacts on native biodiversity or ecosystem functioning; and these impacts are also modified through species by habitat interactions. For instance, Hughes *et al.* (2005) found for nitrogen-deficient habitat on young volcanic flows in Hawaii that soil nitrogen availability was up to 121-times higher in stands invaded by the alien nitrogen-fixing tree, *Falcataria moluccana* compared to native stands; and they also document as a result of the invasion an increase in phosphorus availability, and invasion rates of other alien species such as *Psidium cattleianum.* In contrast, on very old and highly weathered phosphorus-poor soils in the Seychelles the same species had no positive impact on soil nitrogen or phosphorus availabilities or the growth rates of juveniles of other alien species (Kueffer *et al.*, 2008; Kueffer, 2010a).

To improve existing invasive species risk screening systems for oceanic islands (e.g. Daehler *et al.*, 2004) it will be promising to identify risk factors separately for different invasion phases and habitat types based on comparisons of plant invasions from many different islands globally. This will require that the data that is currently store in different regional invasive plant databases is standardised, combined and complemented.

IMPACTS OF ALIEN RATS ON NATIVE FLORA AND FAUNA

Alien rats are present on most oceanic islands around the world where they often have devastating impacts on the native fauna and flora (Blackburn *et al.,* 2004; Jones *et al.,* 2008; Caujapé-Castells *et al.*, 2010). Eradication of rats is possible from small islands but not from larger islands or those with a significant human population (Howald *et al.*, 2007). Much island biodiversity will therefore in the future only survive in the wild if the native species can coexist with rats; possibly thanks to management interventions that mitigate the impacts of rats. This requires a thorough understanding of the impacts of alien rats on native island biodiversity.

General theoretical concepts about food web interactions are crucial for understanding impacts of introduced rats on islands (e.g. Drake & Hunt, 2009; Mulder *et al.,* 2011). Depending on the ecosystem, rats play different functional roles in a food web. They can for instance be predators at different trophic levels (e.g. top predators vs. mesopredators) or competitors of other native or alien predators. Thereby they can be involved in trophic cascades (e.g. as a top predator they may positively affect a native prey species via the reduction of the population density of a mesopredator) or apparent competition (e.g. a native prey species may profit from the predatory effect of rats on a competing native or alien species). Similarly, rats can either have negative impacts on plant regeneration as seed or seedling predators or positive ones as seed dispersers (e.g. Shiels & Drake, 2011).

Among others, empirical generalisation from multiple case studies indicates that the vulnerability of native birds (Jones *et al.*, 2008) or plants (Meyer & Butaud, 2009; Shiels & Drake, 2011) to rat predation differs between species. For instance, small burrow-nesting seabirds are particularly vulnerable to predation by rats, while large ground-nesting birds are less affected (Jones *et al.*, 2008). Or, plant species with small seed tend to be dispersed by rats, while large-seeded species tend to be predated (Shiels & Drake, 2011). Meyer & Butaud (2009) further suggest that some families, such as Arecaceae, Elaeocarpaceae, Rubiaceae, Santalaceae, and Sapotaceae, are vulnerable to seed predation, while other families with soft barks such as Araliaceae, Euphorbiaceae, and Malvaceae suffer particularly from stem or bark damages. Other relevant factors include for instance seasonality (e.g. seedling predation can be greater in the dry season, Caujapé-Castells *et al.,* 2010), habitat context (e.g. bird nest predation is reduced in alien Cryptomeria forests in Mauritius, Safford & Jones, 1998), or history (e.g. those bird species that are most susceptible to predation are already extinct, Blackburn *et al.*, 2004).

Over recent years a large data and knowledge base on impacts of rats (as well as of other mammals) on island faunas and floras accumulated (e.g. Drake & Hunt, 2009; Mulder *et al.,* 2011); but a systematic synthesis is missing. A better understanding of rat impacts on island biodiversity is the kind of problems that is ripe for generalisation through systematic cross-island learning.

Kaiser-Bunbury *et al.* (2010) discuss potentials for comparative research on plant-animal interactions across oceanic islands. Plant-animal mutualisms are fundamental for the fitness of both the animal and plant partners, but they are heavily disturbed on islands. Ambitious 'rewilding' projects have been proposed that aim at restoring ecological interactions by introducing alien analog species of extinct native species; e.g. Aldabran giant tortoise to replace extinct giant tortoises in Mauritius and Rodrigues (Hansen *et al.,* 2010; Kaiser-Bunbury *et al.*, 2010). Alien plant and animal species have strong positive and negative effects on both individual interactions and the structure of mutualistic networks (Olesen *et al.,* 2002; Bascompte & Jordano, 2007; Caujapé-Castells *et al.*, 2010; Kaiser-Bunbury *et al.*, 2010), and invasive species control programs must therefore carefully consider the potential impacts on ecological interactions. An in-depth understanding of plant-animal interactions on a community level is central for successful conservation of island biodiversity (Kaiser-Bunbury *et al.*, 2010), but the collection of such extensive data is particularly labour-intense (e.g. Kaiser-Bunbury *et al.*, 2009*,* 2011). Generalising insights across case studies from multiple islands is thus an economic necessity.

One important aspect of plant-animal interactions that needs to be understood is the relative importance of positive and negative effects of abundant plant or animal species on co-occurring species that depend on the same mutualistic interactions. An abundant alien plant species for instance can have both positive and negative impacts on co-occurring plants. It may compete for pollinator or seed dispersal services and negatively affect reproduction of neighbouring plants (Traveset & Richardson, 2006), but it can also contribute to increased abundances of mutualists by attracting them to a site or supporting a higher population density through increased food supply (Morales & Traveset, 2009). Consequently the effects of one species on another species via mutualistic interactions strongly depend on the biotic composition of a habitat and differ for instance between unrestored and restored sites or along gradients from native to alien dominated vegetation (Bartomeus *et al.,* 2008; Kaiser-Bunbury *et al.*, 2009, 2011; Padrón *et al.,* 2009).

To develop a general ecological theory of relevance to the restoration of plant-animal interactions is likely a major challenge, but generalisations for island ecosystems may be easier to achieve, because these systems are relatively simple and share many characteristics (Kaiser-Bunbury *et al.*, 2010). For instance, pollination and seed dispersal interactions on islands are often highly generalised (i.e. one plant species interacting with many animal mutualists and vice versa). There are typically a few native super-generalists in a mutualistic network that are involved in a large proportion of the possible interactions. A general feature of seed dispersal on islands is that birds, fruit-bats and lizards often make up a major proportion of the disperser community, and consequently a majority of plants produce fleshy fruits. Mutualistic networks on different islands are also faced with the same types of anthropogenic disturbances (loss of native mutualists, introduction of alien species that integrate into mutualistic networks, and habitat fragmentation). Important alien species such as the honey bee (*Apis mellifera*) or the common myna (*Acridotheres tristis*) occur on many islands and their ecology and impacts can thus be compared across islands. Taxonomic and functional biases in the extinctions and rarity of native species are probably also often shared among islands. For instance large seed dispersers such as giant tortoises are extinct on many islands. Taken together, systematic comparisons of mutualistic networks and their restoration across islands are currently lacking, but such cross-island syntheses would be highly relevant for research and management (Kaiser-Bunbury *et al.*, 2010).

MANAGEMENT AND RESTORATION OF ISLAND HABITATS

Island ecosystems are heavily disturbed and island biodiversity is increasingly dependent on continuous conservation management, but historic or 'natural' ecosystem states are not a reliable guide for habitat management of these 'novel' ecosystems anymore (Hobbs *et al.*, 2006; Kueffer & Daehler, 2009). Island conservationists are thus moving through try and error lear-

ning into unmapped terrain, and this requires comparative learning across islands. Many islands share similar habitats with similar anthropogenic disturbances history, which provides an opportunity for such comparisons. For instance, (sub)tropical islands have habitats such as lowland dry habitat, moist to wet mid-elevation forest, montane cloud forest, and subalpine heathlands in common (Mueller-Dombois, 2002). Given these similarities it is surprising that systematic comparisons of habitat management and restoration approaches across islands are rare.

Both in Special Ecological Areas (SEA) in Hawaii (Tunison & Stone, 1992; Loh & Tunison, 2009) and Conservation Management Areas (CMA) in Mauritius (Cheke & Hume, 2008; Florens *et al.*, 2010; Baider & Florens, 2011) wet tropical forest is being conserved and restored through intensive interventions including fencing out of alien animals and complete weeding of alien plants. In contrast, on other islands including Puerto Rico (Lugo, 2004) and Seychelles (Kueffer *et al.*, 2010b) a different approach of facilitating secondary succession of mixed forests composed of both alien and native species has been proposed. Also practical experiences related to more specific aspects of habitat management are accumulating across islands; for instance on *ex situ* propagation and reintroduction to the wild of native plants (e.g. Atkinson *et al.,* 2009), the role of biological control (Meyer & Fourdrigniez, 2011) or community-based approaches. Local communities have for instance been successfully involved in habitat restoration programs on Rodrigues in the Western Indian Ocean (Mauremootoo & Payendee, 2002; Waterstone, 2010), Pitcairn in the Pacific (Warren *et al.*, 2010), or Maui in the Hawaiian archipelago (www.auwahi.org).

To my knowledge few systematic efforts have been made to systematically review and assess habitat management efforts across multiple sites or islands. The toolbox of systematic reviews of conservation evidence may be a useful starting point for more systematic learning (Pullin & Stewart, 2006, www.environmentalevidence.org).

CONCLUSIONS

I have in this article argued that more collaborative research and learning efforts are needed among conservationists from different oceanic islands. Cross-island learning is more than an exchange of facts. It is a way of collaborative thinking and theory formation that depends on long-term continuous interactions (Kueffer, 2006). Multi-year cross-site research projects such as the Seabird Islands and Introduced Predators (SEAPRE) project on the impacts of introduced predators on seabird islands (Mulder *et al.*, 2011, www.seapre.uaf.edu), or the Mountain Invasion Research Network (MIREN) project on the risk of plant invasions into high elevation ecosystems (including on islands) (Dietz *et al.*, 2006, www.miren.ethz.ch) illustrate the potentials of such intensive collaborations. The recently established Global Island Plant Conservation Network (GIPCN, www.bgci.org/ourwork/islands) may become another useful institution for collaborative activities among island conservationists.

Unique island biodiversity is rapidly disappearing. Island conservationists are confronted with the taunting task of devising and implementing new conservation strategies that address at once and in a very short time span many different interacting and rapidly changing threat factors, which affect thousands of threatened island species. To save much island biodiversity from extinction a global and concerted learning and action effort is urgently needed.

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