Scaling Up Restoration Efforts in the Pacific Islands: A Call for Clear Management Objectives, Targeted Research to Minimize Uncertainty, and Innovative Solutions to a Wicked Problem[1]

Melissa R. Price[2,4] *and Robert J. Toonen*[3]

Abstract: Today, restoration of resilient native ecosystems that minimize extinction risk and maximize ecosystem services has never been more important. Success stories, such as that of Hakalau Forest National Wildlife Refuge and papers in this special feature, demonstrate the value in large-scale efforts. However, the number of decision makers involved and siloed agencies responsible for land and nearshore coastal management, incomplete or contradictory knowledge regarding community and ecosystem dynamics, the large costs associated with complete restoration, and the interconnectedness of restoration decisions with social, cultural, policy, and economic dimensions, clearly classify the process of attempting large-scale restoration of degraded systems as a wicked problem. We wish to outline three focal areas that specifically address each of the challenges associated with wicked problems and that may improve the likelihood of achieving restoration goals. First, managers and decision makers must identify clear objectives that will guide decisions regarding restoration actions. Second, research funded through conservation initiatives should be aimed at reducing uncertainty in ways that increase the probability of choosing a set of management actions likely to have a desirable outcome. Finally, we need innovative solutions that borrow from industries that have already discovered economy of scale, as well as partnerships among those in the fields of social science, economics, policy, and the natural sciences. The collection of articles in this special feature illustrate how objective-driven studies, high-value research in areas of uncertainty, and collaborations among economists, cultural practitioners, and scientists can move us toward the identification of optimal solutions. Ultimately, we wish to restore systems in ways that reduce conservation reliance and result in resilient, self-sustaining native ecosystems.

The Pacific islands face ongoing threats to native ecosystems, including introduced predators, pests, disease, wildfire, and a loss of suitable habitat due to human development, climate change, and sea level rise (Arcilla et al. 2015, Taylor and Kumar 2016, Cowie et al. 2017). Due to the massive number of extinctions since human arrival, gaps in native mutualisms among plants, pollinators, and seed dispersers contribute to ongoing population declines in native species (Hanna et al. 2013, Miller et al. 2015, Pejchar 2015).

Less than half of the landscape in Hawai'i is currently covered by native vegetation (Gon et al. 2006), and this is unlikely to change without intentional, intensive restoration efforts. Ecosystems dominated by nonnative species, in many cases intentionally introduced to reduce erosion and improve groundwater recharge, are often resistant to

[1] Manuscript accepted 15 May 2017.

[2] Department of Natural Resources and Environmental Management, College of Tropical Agriculture and Human Resources, University of Hawai'i at Mānoa, Honolulu, Hawai'i 96822.

[3] Hawai'i Institute of Marine Biology, School of Ocean and Earth Sciences and Technology, University of Hawai'i at Mānoa, Kāne'ohe, Hawai'i 96822.

[4] Corresponding author (e-mail: pricemel@hawaii.edu).

Pacific Science (2017), vol. 71, no. 4:391–399
doi:10.2984/71.4.1

recolonization by native species (Ostertag et al. 2008). Additional novel pests and diseases arrive in the Islands each year as hitchhikers on shipped goods and human travelers (Hulme 2009). Wildfires, exacerbated by drought conditions, are also increasing the rate of conversion from native- to nonnative-dominated systems (Trauernicht et al. 2015).

Today, the restoration of resilient native ecosystems that minimize extinction risk and maximize ecosystem services has never been more important. Given the accelerated loss, the best defense of native ecosystems in Hawai'i may be a good offense. We must work to restore native landscapes in places dominated by invasive vegetation and introduced wildlife. Success stories, such as those of Hakalau Forest National Wildlife Refuge (Camp et al. 2010) and papers in this special feature (Judge et al. 2017, Kurashima et al. 2017, Powell et al. 2017, Rowe et al. 2017, Wada et al. 2017, Winter and Lucas 2017), demonstrate the value in large-scale efforts. Novel landscape management approaches may now be used to restore native vegetation at large scales on deserted ranchlands, abandoned agricultural landscapes, and nonnative-dominated forests (Friday et al. 2015). Efforts at restoration must also be scaled up in wetlands, estuaries, and marine habitats such as coral reefs and seagrass beds to create resiliency throughout landscapes from mountain ridges to the nearshore waters receiving those land-based inputs (mauka to makai).

The large number of decision makers involved and agencies responsible for land and nearshore coastal management, incomplete or contradictory knowledge regarding community and ecosystem dynamics, the large costs associated with complete restoration, and the interconnectedness of restoration decisions with social, cultural, policy, and economic dimensions clearly classify the process of attempting large-scale restoration of degraded systems as a wicked problem (Rittel and Webber 1973). In this special collection of articles featured in *Pacific Science*, we invited authors to submit manuscripts addressing restoration of native ecosystems at a landscape scale.

Although we do not pretend to offer silver bullets to solve challenges facing restoration efforts in the Pacific, we wish to outline three focal areas that specifically address each of the challenges associated with wicked problems and that may improve the likelihood of achieving restoration goals. First, managers and decision makers must identify clear objectives that will guide decisions regarding restoration actions. Second, research funded through conservation initiatives should be aimed at reducing uncertainty in ways that increase the probability of choosing a set of management actions likely to have a desirable outcome. Finally, we need innovative solutions that borrow from industries that have already discovered economy of scale, as well as partnerships among researchers and practitioners in the fields of social science, economics, policy, and the natural sciences.

Clearly Defined Objectives

Fundamental objectives, often based on societal values or policy mandates, govern the choices of decision makers (Keeney 1992). Transparency in setting fundamental objectives may improve accountability, public trust, and efficiency in identifying optimal solutions (Elliott and Resnik 2014). Fundamental objectives for state and federal land managers are likely to include minimizing costs, minimizing extinction risk for native species, maximizing groundwater recharge, and preserving access to and ecological integrity for cultural sites. Engaging stakeholders in the decision-making process can minimize conflict and increase support for management actions, particularly when stakeholder objectives are incorporated alongside decision-maker objectives (Ananda and Herath 2003). Stakeholder objectives might include maximizing recreational access to designated lands, maximizing opportunities for cultural practice and education, maximizing agricultural production on designated lands, or maximizing access to subsistence resources such as game or forestry species or fisheries.

The ways in which we achieve fundamental objectives, referred to as means objectives, are often confused with fundamental objectives

(Keeney 1992). For example, the removal of invasive species is often seen as a fundamental objective. This focus on a threat, rather than on the fundamental objectives that motivate removal of invasive species, may distract from the identification of the most efficient, cost-effective solutions. Once decision makers, in consultation with stakeholders, have identified fundamental objectives, then all potential ways of achieving the fundamental objectives can be identified and weighed against one another. For example, removal of invasive species, soil amendments, outplanting, and fencing are all potential ways in which we might reduce extinction risk for native species. Clearly discriminating between fundamental and means objectives increases the likelihood that decision makers will choose solutions that are in line with policy and values.

Some threats are large enough that they are treated as fundamental objectives. For example, minimizing the risk of wildfire is a fundamental objective for many decision makers. The occurrence of wildfire is increasing in the Pacific islands (Trauernicht et al. 2015). In past decades wildfires mostly occurred in lowland agricultural areas, or in grassland, or dry forests. Recently, however, wildfires have begun to push into the periphery of wet native forest habitat. Wildfires may directly cause mortality of native species and may lead to extinctions if they occur in areas with rare species. However, they also offer opportunities for replanting with native plants, as wildfires remove invasive species, and may in some cases favor native plant species (Daehler and Goergen 2005). In Wada et al. (2017), the authors present a spatial assessment of costs associated with restoring habitat to achieve two different objectives, minimizing fire risk and maximizing groundwater recharge.

Competing management objectives often exist and must be factored into restoration decisions. For example, invasive slugs negatively impact outplanting success and reduce seedling survival (Joe and Daehler 2008). However, in areas where endangered native mollusks occur, molluscicide use to protect native plants would be counterproductive to the goal of minimizing extinction risk for endangered native snails. State decision makers in Hawai'i are tasked with managing feral pigs as both a game species, where the fundamental objective is to maximize harvest, and as an invasive species, where the management objective is to minimize impacts of feral pigs on native species. Likewise use preferences may differ, with some wanting access for wildlife harvest, whereas others may wish to conserve those same species, or the site may be sacred (or otherwise kapu) for cultural reasons that would be at odds with increased access for recreation and harvest. The likely existence of competing management objectives highlights the importance of engaging all stakeholders in the identification of both fundamental and means objectives (Failing et al. 2013).

Minimizing and Accounting for Uncertainty

Once all means objectives, or potential solutions, have been identified, decision makers must weigh out the probability that each potential solution has of achieving the prioritized fundamental objectives (Keeney 1992). However, there are often unknowns that produce uncertainty regarding the efficacy of various solutions (Regan et al. 2002, Hildebrandt and Knoke 2011, Yemshanov et al. 2012). For example, there is a high degree of uncertainty associated with climate change in restoration decisions (Yousefpour et al. 2012). In some cases, expert knowledge may be elicited to provide the likely outcome of various management actions (Martin et al. 2012). In other cases, research can reduce uncertainty by providing data regarding the range of improvement a particular solution is likely to achieve. For example, Hawaiian honeycreepers are threatened by both predation and avian malaria, and the magnitude of these threats is influenced by climate change (Rock et al. 2012). If the proportion of the population resistant to avian malaria is known, along with the likelihood of nonresistant individuals contracting the disease, one can weigh the solutions associated with disease control against solutions associated with predator control. Thus, the question decision makers must ask before tasking researchers is whether the reduction in uncertainty regarding a particular

solution is likely to change their decision in a way that increases the probability of achieving the fundamental objectives.

In the last 30 years, progress has been made in reducing uncertainty in the field of restoration ecology by identifying effective approaches to restoration, particularly in forest and grassland ecosystems. The complete eradication of introduced ungulates, rodents, or habitat-altering plants has been accomplished on a few small islands and within conservation fencing (Hess 2016, Judge et al. 2017). However, Judge et al. (2017) are quick to point out that the effort was substantial and costly, and there are considerable difficulties in scaling up to eradication efforts for entire populations on unfenced landscapes. Research on abiotic factors potentially influencing restoration efforts, such as nutrient cycling, light availability, and hydrology (Michaud et al. 2015), has reduced uncertainty regarding optimal restoration techniques.

Ungulate-exclusion fencing, targeted at excluding feral pigs, goats, mouflon sheep, or axis deer, depending on the island, and removal of cattle, has emerged as a large-scale first step in protection and restoration of native ecosystems (Leopold and Hess 2017). Benefits of removing ungulates include increasing biomass of microarthropod communities, important to soil decomposition (Vtorov 1993); reductions in native plant and bird mortality due to rooting or foraging; habitat improvement for endangered species (Banko et al. 2014); a reduction in sedimentation and soil loss (Barrios-Garcia and Ballari 2012); and a reduction in transport of invasive seeds and disease (Nogueira-Filho et al. 2009, Samuel et al. 2011). In the special feature presented in this issue, Judge et al. (2017) present a detailed account of the effort required to eradicate ungulates from large fenced areas, and Kurashima et al. (2017) and Winter and Lucas (2017) highlight the importance of a historical ecological approach to ensure that biocultural restoration efforts have community buy-in and support.

Exclusion and removal of ungulates alone does not result in a native species–dominated system (Cabin et al. 2000, Scowcroft 2013) and may actually promote water runoff and sediment loss in some cases (Strauch et al. 2016). Removal of invasive grasses or other dominant invasive plant species, as well as intentional outplanting or seeding, are often necessary to allow establishment of native plant communities (Cabin et al. 2000, Ammondt et al. 2013, Pinto et al. 2015, Leopold and Hess 2017). The amount of weeding required may differ among sites (Cabin et al. 2002) or over time as nutrient dynamics shift (Yelenik and D'Antonio 2013). Powell et al. (2017), in this issue, report a case study highlighting that, despite the high economic costs, ecological restoration can be achieved, and restoration costs decline with economies of scale, arguing that efforts at larger scales and longer time scales will be most cost-effective.

Some conditions appear to favor natural regeneration or expansion of native vegetation and thus may greatly reduce the cost of restoration. Passive restoration can take place through root suckering, as nearby koa (*Acacia koa*) stands expand to fill gaps between planted stands (Scowcroft and Yeh 2013). Once competing nonnative species are removed and conditions are improved, germination from native seed banks may occur (Medeiros et al. 2014), but all plants do not serve equally well as nurse plants (Yelenik et al. 2015). Interactions among nitrogen fixation and uptake by native and nonnative plants may promote nonnative grasses and discourage germination of native plants (Yelenik 2016). In contrast, some native plants, such as ʻōhiʻa lehua (*Metrosideros polymorpha*), promote germination and growth of native plants, potentially by facilitating appropriate microclimates and soil conditions and minimizing competition with nonnative grasses (Yelenik 2016).

In addition to the removal of invasive plants and animals, targeted restoration of mutualistic relationships may be necessary for successful and persistent restoration. For example, plant communities may need intentional efforts to replace native pollinators or seed dispersers that are now extinct (Hanna et al. 2013, Pejchar 2015) or to protect the (often declining) populations of remaining

native pollinators (Miller et al. 2015). In other cases, habitat restoration to conserve native animals may restore mutualistic relationships such as those between native plants, which provide nesting habitat, and seabirds, which deposit nitrogen into the system (Spatz et al. 2014, VanderWerf et al. 2014). For example, Rowe et al. (2017), in this issue, trace the flow of nutrients from seabirds through a forest ecosystem and show that nearly one-third of foliar nitrogen comes from marine sources (i.e., seabird feces). The montane forest seabirds in the Hawaiian Islands have been greatly reduced, both in number and in range, yet even in vastly reduced numbers, these birds impact soil and vegetation nutrient content (Rowe et al. 2017). Thus it is likely that seabirds in the Hawaiian Islands historically played a major role in controlling soil fertility in areas where they nested, and understanding and compensating for this decreased nutrient flow is vital to efforts to rebuild the ecosystem and restore endemic plants and wildlife (Rowe et al. 2017).

Innovative Partnerships for Sustainable Solutions

Decision makers often lean toward solutions that are familiar, excluding novel ideas that could potentially be more effective. However, wicked problems such as restoration are defined by the interconnected nature of challenging social, cultural, economic, and policy dimensions (Rittel and Webber 1973). Innovative solutions are thus most likely to emerge from transdisciplinary partnerships between those who have already discovered economy of scale and those in the fields of social sciences, economics, policy, and the natural sciences. Several articles in this special feature address cultural, historical, and economic dimensions of restoration (Kurashima et al. 2017, Powell et al. 2017, Wada et al. 2017, Winter and Lucas 2017).

Community engagement in restoration efforts may have unexpected benefits, including cultural benefits (Kittinger et al. 2016, Kurashima et al. 2017). Volunteer labor buoys many restoration efforts, providing the manual labor for pulling weeds and outplanting native plants that otherwise would inflate restoration costs (Holl and Howarth 2000). Winter and Lucas (2017), in this issue, point out that "comanagement efforts that take a biocultural approach (being aligned with community priorities and founded in cultural values) can increase community engagement and thus garner more support for conservation efforts than ones that exclude communities and indigenous cultural perspectives. With the onslaught of invasive species and the impacts of global climate change, large-scale conservation and restoration efforts need to utilize new tools, but that does not mean that these efforts need to reinvent the wheel. Long-term success of conservation efforts is more likely when they are built off of an engaged and supportive local community."

Restoration goals may be achieved through economic incentives for private landowners, particularly for *Acacia koa* forests (Goldman et al. 2008). The potential for profits from native timber production on former cattle ranching and agricultural lands is increasing as beef production costs have risen and farmers and ranchers look to diversify (Cox and Bredhoff 2003, Wilkinson and Elevitch 2003, Perroy et al. 2016). Other potential incentives include cost sharing via the Hawai'i Forest Stewardship Program, food security and production, and limited grazing of cattle in reforested areas (Goldstein et al. 2008).

However, depending on the management objective (i.e., maximize landowner profit, minimize extinction risk for native species, increase groundwater recharge, etc.) the return-on-investment may be nonlinear over time (Goldstein et al. 2008) and vary under future climate change scenarios as highlighted by Wada et al. (2017). The initial investment in restoration may be high, with low maintenance costs, resulting in a favorable long-term return on investment for ecosystem services such as groundwater recharge (Burnett et al. 2017), as native and restored forests conserve more water than forests dominated by nonnative species (Kagawa et al. 2009, Cavaleri et al. 2014, Hata et al. 2015).

Furthermore, as noted by Wada et al. (2017) and Powell et al. (2017), in this issue, different objectives might be met by different strategies or might vary spatially or tempo-

rally in the likelihood of meeting various objectives.

Conclusions

Wicked problems, such as those associated with attempting large-scale restoration of degraded systems, require clear identification of objectives, targeted reduction in uncertainties, and multidisciplinary collaborations to identify optimal solutions. Ultimately, we wish to restore systems in ways that reduce conservation reliance and result in resilient, self-sustaining native ecosystems (VanderWerf 2009, Reed et al. 2012, Shiels et al. 2014). However, many potential management actions, such as propagation of native plants, production of native seeds, and cost-effective removal of invasive plants and animals at a landscape scale, still await entrepreneurial solutions (Lamb et al. 2005, Friday et al. 2015). Likewise, solutions for ecosystems that link terrestrial and marine systems on Pacific islands, such as aquatic and estuarine systems, have received less attention than forest and coral ecosystems, and a high degree of uncertainty regarding optimal management approaches remains (Englund 2008, Holitzki et al. 2013, Dudley et al. 2014). However, efforts are beginning in these areas, and the improvements to these systems will have positive, cascading effects from the mountains to the sea. This special collection of articles illustrates how objective-driven studies, high-value research in areas of uncertainty, and collaborations among economists, cultural practitioners, and scientists can move us toward the identification of optimal solutions.

Literature Cited

Ammondt, S. A., C. M. Litton, L. M. Ellsworth, and J. K. Leary. 2013. Appl. Veg. Sci. 16:29–39. Restoration of native plant communities in a Hawaiian dry lowland ecosystem dominated by the invasive grass *Megathyrsus maximus*. Appl. Veg. Sci. 16:29–39.

Ananda, J., and G. Herath. 2003. The use of analytic hierarchy process to incorporate stakeholder preferences into regional forest planning. For. Policy Econ. 5:13–26.

Arcilla, N., C. Y. Choi, K. Ozaki, and C. A. Lepczyk. 2015. Invasive species and Pacific island bird conservation: A selective review of recent research featuring case studies of Swinhoe's storm petrel and the Okinawa and Guam rail. Ornithology 156:S199–S207.

Banko, P. C., S. C. Hess, P. G. Scowcroft, C. Farmer, J. D. Jacobi, R. M. Stevens, R. J. Camp, D. L. Leonard Jr., K. W. Brinck, J. O. Juvik, and S. P. Juvik. 2014. Evaluating the long-term management of introduced ungulates to protect the Palila, an endangered bird, and its critical habitat in subalpine forest of Mauna Kea, Hawai'i. Arct. Antarct. Alp. Res. 46:871–889.

Barrios-Garcia, M. N., and S. A. Ballari. 2012. Impacts of wild boar (*Sus scrofa*) in its introduced and native range: A review. Biol. Invasions 14:2283–2300.

Burnett, K., C. Wada, and A. Balderston. 2017. Benefit-cost analysis of watershed conservation on Hawai'i Island. Ecol. Econ. 131:262–274.

Cabin, R. J., S. G. Weller, D. H. Lorence, S. Cordell, L. J. Hadway, R. Montgomery, D. Goo, and A. Urakami. 2002. Effects of light, alien grass, and native species additions on Hawaiian dry forest restoration. Ecol. Appl. 12:1595–1610.

Cabin, R. J., S. G. Weller, D. H. Lorence, T. W. Flynn, A. K. Sakai, D. Sandquist, and L. J. Hadway. 2000. Effects of long-term ungulate exclusion and recent alien species control on the preservation and restoration of a Hawaiian tropical dry forest. Conserv. Biol. 14:439–453.

Camp, R. J., T. K. Pratt, P. M. Gorresen, J. J. Jeffrey, and B. L. Woodworth. 2010. Population trends of forest birds at Hakalau Forest National Wildlife Refuge, Hawai'i. Condor 112:196–212.

Cavaleri, M. A., R. Ostertag, S. Cordell, and L. Sack. 2014. Native trees show conservative water use relative to invasive trees: Results from a removal experiment in a Hawaiian wet forest. Conserv. Physiol. 2:1–14.

Cowie, R. H., C. Regnier, B. Fontaine, and P. Bouchet. 2017. Measuring the sixth extinction: What do mollusks tell us? Nautilus 1:3–41.

Cox, L. J., and S. Bredhoff. 2003. The Hawai'i beef industry: Situation and outlook update. Livestock Management Series LM-8, College of Tropical Agriculture and Human Resources, University of Hawai'i at Mānoa, Honolulu.

Daehler, C. C., and E. M. Goergen. 2005. Experimental restoration of an indigenous Hawaiian grassland after invasion by buffel grass (*Cenchrus ciliaris*). Restor. Ecol. 13:380–389.

Dudley, B. D., R. A. MacKenzie, T. S. Sakihara, H. Dulaiova, C. A. Waters, R. F. Hughes, and R. Ostertag. 2014. Influences of N-fixing and non-N-fixing vegetation and invasive fish on water chemistry of Hawaiian anchialine ponds. Pac. Sci. 68:509–523.

Elliott, K. C., and D. B. Resnik. 2014. Science, policy, and the transparency of values. Environ. Health Perspect. 122:647–650.

Englund, R. A. 2008. Invasive species threats to native aquatic insect biodiversity and conservation measures in Hawai'i and French Polynesia. J. Insect Conserv. 12:415–428.

Failing, L., R. Gregory, and P. Higgins. 2013. Science, uncertainty, and values in ecological restoration: A case study in structured decision-making and adaptive management. Restor. Ecol. 21:422–430.

Friday, J. B., S. Cordell, C. P. Giardina, F. Inman-Narahari, N. Koch, J. K. Leary, C. M. Litton, and C. Trauernicht. 2015. Future directions for forest restoration in Hawai'i. New For. 46:733–746.

Goldman, R. L., L. Pejchar Goldstein, and G. C. Daily. 2008. Assessing the conservation value of a human-dominated island landscape: Plant diversity in Hawai'i. Biodivers. Conserv. 17:1765–1781.

Goldstein, J. H., L. Pejchar, and G. C. Daily. 2008. Using return-on-investment to guide restoration: A case study from Hawai'i. Conserv. Lett. 1:236–243.

Gon, S., A. Allison, R. Cannarella, J. Jacobi, K. Kaneshiro, M. Kido, M. Lane-Kamahele, and S. Miller. 2006. A GAP analysis of Hawai'i: Final report. U.S. Geological Survey, Washington, D.C.

Hanna, C., D. Foote, and C. Kremen. 2013. Invasive species management restores a plant-pollinator mutualism in Hawai'i. Appl. Ecol. 50:147–155.

Hata, K., K. Kawakami, and N. Kachi. 2015. Higher soil water availability after removal of a dominant, nonnative tree (*Casuarina equisetifolia* Forst.) from a subtropical forest. Pac. Sci. 69:445–460.

Hess, S. 2016. A tour de force by Hawaii's invasive mammals: Establishment, takeover, and ecosystem restoration through eradication. Mammal Study 41:47–60.

Hildebrandt, P., and T. Knoke. 2011. Investment decisions under uncertainty: A methodological review on forest science studies. For. Policy Econ. 13:1–15.

Holitzki, T. M., R. A. MacKenzie, T. N. Wiegner, and K. J. McDermid. 2013. Differences in ecological structure, function, and native species abundance between native and invaded Hawaiian streams. Ecol. Appl. 23:1367–1383.

Holl, K., and R. B. Howarth. 2000. Paying for restoration. Restor. Ecol. 8:260–267.

Hulme, P. E. 2009. Trade, transport, and trouble: Managing invasive species pathways in an era of globalization. Appl. Ecol. 46:10–18.

Joe, S. M., and C. C. Daehler. 2008. Invasive slugs as under-appreciated obstacles to rare plant restoration: Evidence from the Hawaiian Islands. Biol. Invasions 10:245–255.

Judge, S. W., S. C. Hess, J. K. Faford, D. Pacheco, and C. R. Leopold. 2017. Monitoring eradication of European mouflon sheep from the Kahuku Unit of Hawai'i Volcanoes National Park. Pac. Sci. 71:425–436.

Kagawa, A., L. Sack, K. Duarte, and S. James. 2009. Hawaiian native forest conserves water relative to timber plantation: Species and stand traits influence water use. Ecol. Appl. 19:1429–1443.

Keeney, R. L. 1992. Value-focused thinking: A path to creative decision-making. Harvard University Press, Cambridge, Massachusetts.

Kittinger, J. N., T. M. Bambico, D. Minton, A. Miller, M. Mehia, N. Kalei, B. Wong, and E. W. Glazier. 2016. Restoring ecosystems, restoring community: Socioeconomic and cultural dimensions of a community-based coral reef restoration project. Reg. Environ. Change 16:301–313.

Kurashima, N., J. Jeremiah, and T. Ticktin. 2017. I ka wā ma mua: The value of a historical ecology approach to ecological restoration in Hawai'i. Pac. Sci. 71:437–456.

Lamb, D., P. D. Erskine, and J. A. Parrotta. 2005. Restoration of degraded tropical forest landscapes. Science (Washington, D.C.) 310:1628–1632.

Leopold, C. R., and S. C. Hess. 2017. Conversion of native terrestrial ecosystems in Hawai'i to novel grazing systems: A review. Biol. Invasions 19:161–177.

Martin, T. G., M. A. Burgman, F. Fidler, P. M. Kuhnert, S. Low-Choy, M. McBride, and K. Mengersen. 2012. Eliciting expert knowledge in conservation science. Conserv. Biol. 26:29–38.

Medeiros, A. C., E. I. von Allmen, and C. G. Chimera. 2014. Dry forest restoration and unassisted native tree seedling recruitment at Auwahi, Maui. Pac. Sci. 68:33–45.

Michaud, J., S. Cordell, T. C. Cole, and R. Ostertag. 2015. Drought in an invaded Hawaiian lowland wet forest. Pac. Sci. 69:367–383.

Miller, A. E., B. J. Brosi, K. Magnacca, G. C. Daily, and L. Pejchar. 2015. Pollen carried by native and nonnative bees in the large-scale reforestation of pastureland in Hawai'i: Implications for pollination. Pac. Sci. 69:67–79.

Nogueira-Filho, S. L., S. S. Nogueira, and J. M. Fragoso. 2009. Ecological impacts of feral pigs in the Hawaiian Islands. Biodivers. Conserv. 18:3677.

Ostertag, R., C. P. Giardina, and S. Cordell. 2008. Understory colonization of *Eucalyptus* plantations in Hawai'i in relation to light and nutrient levels. Restor. Ecol. 16:475–485.

Pejchar, L. 2015. Introduced birds incompletely replace seed dispersal by a native frugivore. AOB Plants 7:plv072.

Perroy, R. L., J. Melrose, and S. Cares. 2016. The evolving agricultural landscape of post-plantation Hawaii. Appl. Geogr. 76:154–162.

Pinto, J. R., A. S. Davis, J. J. K. Leary, and M. M. Aghai. 2015. Stocktype and grass suppression accelerate the restoration trajectory of *Acacia koa* in Hawaiian montane ecosystems. New For. 46:855–867.

Powell, K. B., L. M. Ellsworth, C. M. Litton, K. L. L. Oleson, and S. A. Ammondt. 2017. Toward cost-effective restoration: Scaling up restoration in ecosystems degraded by nonnative invasive grass and ungulates. Pac. Sci. 71:479–493.

Reed, J. M., D. W. Desrochers, E. A. VanderWerf, and J. M. Scott. 2012. Long-term persistence of Hawaii's endangered avifauna through conservation-reliant management. BioScience 62:881–892.

Regan, H. M., M. Colyvan, and M. A. Burgman. 2002. A taxonomy and treatment of uncertainty for ecology and conservation biology. Ecol. Appl. 12:618–628.

Rittel, H. W. J., and M. M. Webber. 1973. Dilemmas in a general theory of planning. Policy Sci. 4:155–169.

Rock, K. S., M. J. Keeling, and D. A. Wood. 2012. Modelling the future of the Hawaiian honeycreeper: An ecological and epidemiological problem. Ecol. Modell. 235–236:26–35.

Rowe, J. A., C. M. Litton, C. A. Lepczyk, and B. N. Popp. 2017. Impacts of endangered seabirds on nutrient cycling in montane forest ecosystems of Hawai'i. Pac. Sci. 71:495–509.

Samuel, M. D., P. H. Hobbelen, F. DeCastro, J. A. Ahumada, D. A. LaPointe, C. T. Atkinson, B. L. Woodworth, P. J. Hart, and D. C. Duffy. 2011. The dynamics, transmission, and population impacts of avian malaria in native Hawaiian birds: A modeling approach. Ecol. Appl. 21:2960–2973.

Scowcroft, P. G. 2013. Parent tree effects on reestablishment of *Acacia koa* in abandoned pasture and the influence of initial density

on stand development. New For. 44:409–426.

Scowcroft, P. G., and J. T. Yeh. 2013. Passive restoration augments active restoration in deforested landscapes: The role of root suckering adjacent to planted stands of *Acacia koa*. For. Ecol. Manage. 305:138–145.

Shiels, A. B., W. C. Pitt, R. T. Sugihara, and G. W. Witmer. 2014. Biology and impacts of Pacific island invasive species. 11. *Rattus rattus*, the black rat (Rodentia: Muridae). Pac. Sci. 68:145–184.

Spatz, D. R., K. M. Newton, R. Heinz, B. Tershy, N. D. Holmes, S. H. M. Butchart, and D. A. Croll. 2014. The biogeography of globally threatened seabirds and island conservation opportunities. Conserv. Biol. 28:1282–1290.

Strauch, A. M., G. L. Bruland, R. A. MacKenzie, and C. P. Giardina. 2016. Soil and hydrological responses to wild pig (*Sus scofa*) exclusion from native and strawberry guava (*Psidium cattleianum*)–invaded tropical montane wet forests. Geoderma 279:53–60.

Taylor, S., and L. Kumar. 2016. Global climate change impacts on Pacific islands terrestrial biodiversity: A review. Trop. Conserv. Sci. 9:203–223.

Trauernicht, C., E. Pickett, C. P. Giardina, C. M. Litton, S. Cordell, and A. Beavers. 2015. The contemporary scale and context of wildfire in Hawai'i. Pac. Sci. 69:427–444.

VanderWerf, E. A. 2009. Importance of nest predation by alien rodents and avian poxvirus in conservation of Oahu Elepaio. J. Wildl. Manage. 73:636–746.

VanderWerf, E. A., L. C. Young, S. E. Crow, E. Opie, H. Yamazaki, C. J. Miller, D. G. Anderson, L. S. Brown, D. G. Smith, and J. Eijzenga. 2014. Increase in Wedge-tailed Shearwaters and changes in soil nutrients following removal of alien mammalian predators and nitrogen-fixing plants at Kaena Point, Hawaii. Restor. Ecol. 22:676–684.

Vtorov, I. V. 1993. Feral pig removal: Effects on soil microarthropods in a Hawaiian rain forest. J. Wildl. Manage. 57:875–880.

Wada, C. A., L. L. Bremer, K. Burnett, C. Trauernicht, T. Giambelluca, L. Mandle, E. Parsons, C. Weil, N. Kurashima, and T. Ticktin. 2017. Estimating cost-effectiveness of Hawaiian dry forest restoration using spatial changes in water yield and landscape flammability under climate change. Pac. Sci. 71:401–424.

Wilkinson, K. M., and C. R. Elevitch. 2003. Growing koa: A Hawaiian legacy tree. Permanent Agriculture Resources, Hōlualoa, Hawai'i.

Winter, K. B., and M. Lucas. 2017. Spatial modeling of social-ecological management zones of the ali'i era on the island of Kaua'i, with implications for large-scale biocultural conservation and forest restoration efforts in Hawai'i. Pac. Sci. 71:457–477.

Yelenik, S. G. 2016. Linking dominant Hawaiian tree species to understory development in recovering pastures via impacts on soils and litter. Restor. Ecol. 25:42–52.

Yelenik, S. G., and C. M. D'Antonio. 2013. Self-reinforcing impacts of plant invasions change over time. Nature (Lond.) 503:517–520.

Yelenik, S. G., N. DiManno, and C. M. D'Antonio. 2015. Evaluating nurse plants for restoring native woody species to degraded subtropical woodlands. Ecol. Evol. 5:300–313.

Yemshanov, D., F. H. Koch, D. B. Lyons, M. Ducey, and K. Koehler. 2012. A dominance-based approach to map risks of ecological invasions in the presence of severe uncertainty. Divers. Distrib. 18:33–46.

Yousefpour, R., J. B. Jacobsen, B. J. Thorsen, H. Meilby, M. Hanewinkel, and K. Oehler. 2012. A review of decision-making approaches to handle uncertainty and risk in adaptive forest management under climate change. Ann. For. Sci. 69:1–15.