Chapter 14

FAUNA AND NOVEL ECOSYSTEMS

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The concept of novel ecosystems has largely arisen from the field of restoration ecology, which historically has been dominated by plant ecologists (Majer 2009). Most restoration studies focus on flora with relatively little work conducted on fauna (Lugo et al. 2012); fauna may therefore be overlooked when novel ecosystems are discussed and managed. Conversations about novel ecosystems should include these higher trophic levels and not assume their dynamics are represented by the changes occurring in the abiotic and vegetative realms. This chapter was written to ensure a faunal perspective is integrated into discussions of novel ecosystems. In this chapter, we discuss causes and rates of novel ecosystem formation as they pertain to fauna, faunal use of novel ecosystems and how novel ecosystems can be used to achieve faunal conservation goals.

14.1 WHAT IS A NOVEL ECOSYSTEM FROM A FAUNAL PERSPECTIVE?

From a faunal perspective, current novel ecosystems: (1) contain non-native vegetation and/or animals; (2) contain native species mixes not historically present due to rapid recent range expansions; (3) have altered abundances or interactions partially or wholly mediated by anthropogenic activities; (4) may no longer contain historical ecological engineers or keystone species (*sensu* Daily et al. 1993; Brown 1995); and/or (5) have missing or added trophic levels or other interactions.

There are numerous examples in this book as well as the restoration and invasion literature of novelty as a result of changes in plant communities. These novel plant communities can provide habitat features for

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native fauna and this will be discussed in more detail in Section 14.5.3. Novelty created by introduced animals is also well documented in the literature, particularly when the animal(s) are ecological engineers. Famous examples include the brown tree snake (*Boiga irregularis*) invasion of Micronesia (Savidge 1987; Fritts and Rodda 1998), the restoration of beaver (*Castor* spp) throughout the Holarctic (Rosell et al. 2005) and their subsequent introduction in South America (Anderson et al. 2009), feral populations of agricultural animals that have naturalized throughout the globe (Mack et al. 2000) and the red fox, feral cat and European rabbit introductions in Australia (Dickman 1996). Introductions of these non-native animals have resulted in direct and indirect ecosystem effects that have altered the abiotic characteristics of ecosystems as well as abundances or interactions of native fauna. In many instances these changes are irreversible and result in novel ecosystems (Chapter 3).

Less well studied are novel ecosystems that are being created due to human-facilitated rapid range expansions of native species. The reasons for these expansions are not clear but presumably the taxa are responding positively to rapid and widespread anthropogenic creation of new habitat. The presence of these species can result in new biotic interactions that cause changes in abundance of native species of conservation concern. One of the best examples is the recent expansion of the larger and more aggressive North American barred owl (*Strix varia*), which now co-occurs and out-competes the US federally listed threatened spotted owl (*S. occidentalis*) in restored old-growth forests in the Pacific Northwest (Dugger et al. 2011). Historically, the barred owl was limited to eastern North America. In the early 1900s, its range gradually began to extend westward across wooded regions of central Canada and British Columbia then north into southeast Alaska and south into western Montana and Idaho (Livezey 2009). It was first documented in Washington in 1965 and then in Oregon in 1974 (Kelly et al. 2003; Livezey 2009). Between 1974 and 1998, Kelly et al. (2003) estimated that 706 different barred owl territories were established in Oregon.

The barred owl occupies a wide range of forest conditions, and it has been hypothesized that its range expansion was facilitated by forest management practices that reduced availability of older forests and increased forest fragmentation. Using multi-season occupancy models with a large dataset, Dugger et al. (2011) have clearly documented strong interference competition between the two species. Spotted owl territory occupancy increased when the proportion of old forest increased and/or the degree of fragmentation was decreased. In addition, occupancy rates decreased when barred owls were detected, regardless of the territory habitat configuration. Owl territories where barred owls are detected each year continue to increase, so it is clear that the two species have not yet reached equilibrium and the spotted owl may go extinct before this occurs. It is not clear how this competitive interaction will affect trophic dynamics in these forest ecosystems. This may result only in a species replacement or have cascading effects throughout the food web, resulting in novel assemblages.

Changes in species abundances due to rapid environmental change can be far more dramatic than changes in species distributions. Simpson et al. (2011) make a convincing case that fundamental changes in fish abundances have occurred in the northeast Atlantic during the last 30 years of warming. This investigation was based on detailed analyses of surveys conducted in this well-studied area where sea temperatures have risen by 0.04°C per year in the last three decades. Despite relatively minor changes in the ranges of fishes, the authors found 72% of common fish species have exhibited a significant change in abundance. While some species increased in abundance, others declined. Such changes are likely to have much greater effects on the ecology and exploitation of these fish assemblages than any range extensions or contractions. As noted by Harborne and Mumby (2011), the novel assemblages occurring in the northeast Atlantic are not a simple result of climate change but are likely due to species-specific responses to the warming, changed inter-specific interactions from these changed abundances and the level of exploitation which appears to shift based on changing abundances of harvested species (see fig. 1 in Harborne and Mumby 2011 for an illustration of these processes).

Novelty created by the loss of ecological engineers and/or keystone species is well illustrated by recent studies on the ecological effects of the common global practice of eradicating top native vertebrate predators. There is strong evidence eradication of grey wolves (*Canis lupus lupus*) and dingos (*C. l. dingo*) has influenced distribution and densities of a wide range of species that occupy different trophic levels including mesocarnivores, marsupials, rodents, riparian songbirds and palatable plants (Johnson et al. 2007; Ripple and Beschta 2007; Letnic et al. 2009). Similarly, ant

and spider manipulation experiments reveal a range of effects these groups have on herbivores, decomposers, parasitoids, pollinators, other predators and plants (Lach 2008; Schmidt-Entling and Siegenthaler 2009; Schmitz 2009; Brechbühl et al. 2010; Mellbrand et al. 2010; Piñol et al. 2010; Sanders and van Veen 2011).

Changes in trophic complexity can also be coincident with the creation of a novel ecosystem. For example, the invasive yellow crazy ant, which acts as both predator and mutualist, has greatly altered food web dynamics on Christmas Island, Australia. The ants consume honeydew from sap-sucking insects in the forest canopy, and thereby achieve high abundance and contribute to rainforest dieback (O'Dowd et al. 2003; Abbott and Green 2007). When they are abundant, the ants prey on the endemic red land crab, which previously lacked significant predation pressure. The loss of herbivorous red land crabs results in development of a dense understory of seedlings (O'Dowd et al. 2003) and also facilitates invasion by the giant African land snail (Green et al. 2011). Through its direct and indirect effects, the ant has dramatically changed the functioning of the island's forest ecosystem.

Another example from Canadian lakes shows how the invasion of mid-trophic-level zooplankton *Bythotrephes* spp. greatly changed trophic complexity. *Bythotrephes* caused a decline in herbivorous zooplankton and an increase in predaceous zooplankton, resulting in an overall increase in trophic position of zooplankton and lake herring. Although the shifts are small, because of biomagnification up the food chain they result in a 30% increase in mercury concentration in lake herring which are consumed by lake trout, which are in turn consumed by anglers (Rennie et al. 2011).

There is some evidence that non-native species can compensate for changes in biotic interactions resulting from eradication of native species. The contribution of novel species to the maintenance of ecosystem function is discussed in several chapters in this book (see Chapters 3 and 18). In terms of replacement or displacement of native species, there are numerous studies although the vast majority are plant-focused (e.g. Westman 1990; Ewel and Putz 2004; Kiers et al. 2010). Pattemore and Wilcove (2011) investigated whether compensation by non-native vertebrate pollinators could maintain pollination in the face of extinctions of native vertebrate pollinators. Using a natural experiment (a comparison of pollination at islands with and without the endemic vertebrate flower visitors), they demonstrated that two recently arrived species in New Zealand, the ship rat (*Rattus rattus*) and the silvereye (*Zosterops lateralis*; a passerine bird), partly maintain pollination for three native forest plant species in northern New Zealand. They also demonstrated that without this compensation, these plants would be significantly more pollen-limited. This study provides empirical evidence that two widespread non-native species including one known to have severe negative effects on biodiversity (ship rat) can partially play an important role in maintaining at least one ecosystem function. It should be noted, however, that the three plant species studied have very accessible flowers with copious nectar ideal for attracting vertebrate pollinators. This is in contrast to the tubular flower of the endemic (*Rhabdothamnus solandri*), which has diminished following loss of its native pollinators in this system (Anderson et al. 2011; Pattemore and Wilcove 2011; discussed in more detail in Sections 14.4.1 and 14.4.5).

14.2 NOVEL ANIMAL ASSEMBLAGES ARE COMMON HISTORICALLY

While the concept of novel ecosystems is comparatively new, the existence of these ecosystems is not. As noted by Williams and Jackson (2007), no-analog fossil invertebrate and mammal assemblages are pervasive in Quaternary paloecological records and occur in a range of terrestrial and marine ecosystems (see also Chapter 7). The best-known cases are from the most recent glacial–interglacial transition in North America (Edwards et al. 2005). No-analog communities in North America were most extensive between 12,000 and 17,000 years ago and were most prevalent in Alaska and the interior of eastern North America. The development and disappearance of these communities has been linked to climatic changes in these areas (Williams and Jackson 2007).

In addition, many landscapes characterized by a prolonged human presence such as those in Europe support groups of species that may not have lived together without human presence (Lindenmeyer et al. 2008; Jackson 2012). These systems also do not depend on continued human activity for their maintenance (Hobbs et al. 2006). The role of indigenous cultural activities, including agriculture, land clearance, fire setting and hunting in structuring ecosystems

before Euro American occupation is under active debate (see Ellis 2011 and Jackson 2012 for comprehensive reviews). In spite of the debate, it is clear that human interactions with ecosystems are exceedingly complex and dynamic and we have altered terrestrial and aquatic ecosystems both intentionally and unintentionally for millennia.

The point of this historical perspective is that goals for faunal conservation are primarily described in the context of native or historical ecosystems. Thus, when attempting to improve conditions for animals via restoration or preservation the target or desired conditions are generally defined as 'natural' or within the 'historical range of variability' for that site. The reference for 'natural' conditions is typically one without effects from the current dominant culture. Others extend this definition and include environmental effects caused by the region's indigenous cultures. Thus, setting restoration or management goals to create or maintain animal assemblages that have occurred within the historical range of variability requires a clear definition of the timeframe used for defining 'historical' as well as acknowledging rapidly changing global conditions (see next section for more details).

14.3 RATE OF NOVEL ECOSYSTEM CREATION

Novel ecosystems are being created at a rapid rate due to climate change, land-use practices, land-cover changes and species eradications and invasions. Ellis (2011) estimates nearly one-third of the terrestrial biosphere has been transformed into novel ecosystems and the majority of these transformations have occurred in the past century (see also Chapter 8).

The rate of formation of future novel ecosystems is difficult to predict but Williams et al. (2007) used recent climate change predictions to identify regions projected to develop novel 21st century climates. Depending on the emissions scenario, they predicted 4–39% of the Earth's terrestrial surface would have novel climates by 2100 AD. Stralberg et al. (2009) examined the potential change in California terrestrial breeding bird communities as a result of regional and local climate change. Their model predictions (which were based on region-specific climate and species distribution models) indicate that, by 2070, over half of California could be occupied by novel assemblages of bird species, implying the potential for dramatic community reshuffling and altered patterns of species interactions.

14.4 CAUSES OF NOVELTY

Human agency is the key driver in the formation of novel ecosystems. As described in Chapter 5, human influence can be *onsite* (e.g. plantings, species eradications, land-cover change) or *offsite* (e.g. CO_2 or N emissions). They may be *deliberate* or *inadvertent*, or they can be *deliberate with inadvertent consequences*. They can be *acts of commission*, a purposeful undertaking to achieve a goal, or *acts of omission*, an intentional lack of activity. We illustrate these mechanisms of novel ecosystem creation with brief discussions of species introductions, species eradications, climate change, land-use practices and a combination of drivers. These particular types of human influence were chosen because of their noted effects on fauna.

14.4.1 Species introductions

Species introductions are a frequent means of human agency. A hybrid or novel ecosystem ensues when species introduced to an ecosystem function and interact differently from the resident biota. The literature on biological invasions is rich with examples of invertebrates and vertebrates that have been brought either intentionally or accidentally to a new landscape and have greatly altered its ecology via competition, predation, herbivory, parasitism or mutualism. The European honeybee (*Apis mellifera*) has been introduced to almost every country in the world and, although it is one of our most useful crop pollinators, it also likely displaces native pollinators and in some cases preferentially visits alien plant species, creating 'invader complexes' (Goulson 2003; Morales and Aizen 2006; Schweiger et al. 2010). In Puerto Rico however, this bee is as effective as the most abundant native avian pollinator *Coereba flaveola* in pollinating *Goetzea elegans*, an endangered and self-incompatible endemic tree (Caraballo Ortiz and Santiago Valentín 2011).

Introducing an additional species to biologically control an invader is another kind of intentional introduction that can increase novelty, whether or not it is successful in controlling the target species. A classic

example is the cane toad (*Bufo marinus*) which has been introduced to many regions of the world (particularly the Pacific) for the biological control of agricultural pests. Following the apparent success of the cane toad in eating the beetles that were threatening the sugarcane plantations of Puerto Rico, and the fruitful introductions into Hawaii and the Philippines, there was a strong push for the cane toad to be released in Australia to negate the pests that were ravaging the Queensland cane fields. As a result, toads were collected from Hawaii and approximately 60,000 were released in Australia in the 1930s. The toads became firmly established in Queensland, increasing exponentially in number and extending their range across northern Australia (Shine 2010).

It has since become one of the most intensely studied invasive species in the world. Research on the effect of cane toads on native species (reviewed by Shine 2010 and more recently documented e.g. by Brown et al. 2011; Nelson et al. 2011a, b) indicates the major mechanism of effect is lethal toxic ingestion of toads by frog-eating predators, but the magnitude of effect varies dramatically both spatially and temporally and among predator taxa. No native species have yet gone extinct as a result of toad invasion and many native taxa initially thought to be at risk (e.g. native snakes) are not affected, either because of a physiological ability to tolerate toad toxins or an innate or learned reluctance to consume toads. Some of the species initially severely affected by toad invasion recover within a few decades via aversion learning and longer-term adaptive changes (Phillips and Shine 2004; Phillips et al. 2010). Other indirect trophic effects of the toads have apparently resulted in declines of some native species but increases in others (Shine 2010; Brown et al. 2011).

Accidental introductions can have just as dramatic effects. The varroa mite (*Varroa jacobsoni*) is a major pest of honeybees, and its spread from southeast Asia to almost every country in the world was facilitated by the increase in international trade and travel following World War II. In addition to its direct effects as an ectoparasite, it is associated with many bee viruses (Sammataro et al. 2000), some of which have been implicated in the large-scale deaths of honeybees in North America (Bromenshenk et al. 2010). The four rodent species most commonly inadvertently introduced around the world in the shipping trade are implicated in the extinction of at least 11 insular small mammal species (Harris 2009), and their negative effects on invertebrates are also common, widespread and severe (St Clair 2011).

Conservation translocations are intentional species introductions that are often used to enhance faunal populations. Translocations usually consist of reintroducing native fauna from either wild or captive stock back to their historical range (Armstrong and Seddon 2007). Although this is the ideal, the original habitat may be extensively modified and restoration through reintroduction may not be feasible. In these situations, it might be desirable (and necessary to avoid extinction) to move species outside the natural distribution (Seddon and Soorae 1999), creating novelty in the translocation site.

Alternatively, as noted by Parker et al. (2010), an extension of this approach is to replace extinct species with an appropriate analog species. This approach has merit when the conservation goal is ecological restoration rather than species conservation, but the result is the creation or augmentation of a novel ecosystem. The use of ecological analogs by wildlife biologists is not new. Reintroductions that have successfully used subspecific analogs include the peregrine falcon (*Falco peregrinus*), New Zealand takahe (*Porphyrio mantelli*) and Puerto Rican parrot (*Amazona vittata*). Analogs for congeneric extinct species have also been reintroduced successfully including: (1) yellow-crowned night herons (*Nycticorax violacea*) introduced to Bermuda islands; (2) tundra musk oxen (*Ovibos moschatus*) introduced to the Siberian steppe; and (3) Australian brown quail (*Coturnix ypsilophora*) in New Zealand (Parker et al. 2010).

More controversial recent analog proposals include: (1) Atkinson's (1988) suggestion to introduce a variety of primarily Australian species as ecological replacements for extinct New Zealand wildlife and (2) Donlan et al.'s (2005, 2006) proposal to 'rewild' North America with megafauna replacements from the Pleistocene. These ideas have gained little traction given the uncertainty around the release of any extant introduced species (but see Hansen 2010).

14.4.2 Species eradications

Species eradications are another frequent means of human agency that result in novel ecosystems. These eradications have been deliberate (e.g. removal of predators) or inadvertent (e.g. extinction of Guam avifauna caused by the inadvertent introduction of the brown

tree snake) or they can be deliberate with inadvertent consequences (e.g. trophic cascades from removal of wolves). Clearly there are ecological consequences from species eradications, particularly if the species are ecological engineers or keystone species. There is ecological debate about whether every species is required for stable ecosystem function (e.g. Kareiva and Levin 2003). Our objective is not to participate in this debate but to point out that eradication of novel components with the intent of restoring historical conditions may result in unintended negative consequences. The recent experience in the World Heritage island of Macquarie is a sobering example of the need for ecological understanding of the role of invasive species before expensive management actions are undertaken (Bergstrom et al. 2009; Cadotte 2009). A successful eradication program for introduced feral cats resulted in the ecological release of populations of their primary prey, introduced rabbits. The extensive herbivory and burrowing caused by the overabundant rabbit population threatens to exterminate native Macquarie vegetation, which was not a goal of the feral cat eradication.

Another example of potential negative consequences of the removal of novel components is the recent introduction of the non-native biocontrol agent – the tamarisk beetle (*Diorhabda* spp.) – to eradicate tamarisk (*Tamarix* spp.), a genus of non-native trees that has become a dominant component of riparian woodlands in the southwestern US (Friedman et al. 2005). These riparian woodlands support many species of riparian obligates including the southwestern willow flycatcher (*Empidonax traillii extimus*), listed as an endangered subspecies under the US Endangered Species Act. Recently Paxton et al. (2011) used a combination of literature review and demographic modeling to investigate the potential for positive and negative effects of the loss of riparian vegetation from this control program on riparian birds, with a particular focus on the endangered flycatcher. They concluded that species with restricted distributions that include areas dominated by tamarisk (such as the flycatcher) may be negatively affected both in the short and long term. This is because the birds use the tamarisk: in some areas up to 75% of flycatchers nest in tamarisk and reproductive success associated with these nests is indistinguishable from nests in native trees (Sogge et al. 2008). Also, rate of regeneration and/or restoration of native trees may not be fast enough relative to the rate of tamarisk loss due to the permanently altered hydrological characteristics of these riparian areas.

These examples illustrate the complexities associated with the control or eradication of one non-native species: it will not necessarily result in the desired outcome because species interactions may be altered (van Riper et al. 2008; Schlaepfer et al. 2011). Non-native species can become integral to inter-species relationships to such an extent that their eradication can alter well-functioning communities (Ewel and Putz 2004; Prévot-Julliard et al. 2011).

14.4.3 Climate change

The emission of greenhouse gases causing climate change is an offsite human influence that is inescapable even in the most remote landscapes on the planet. Alterations in temperature and rainfall patterns and increased CO_2 are all part of human-induced global climate change that affect the functioning of and interactions in ecosystems (see Chapter 10). Changes in the phenology of species and their latitudinal and altitudinal distribution with increases in temperature have been documented for many species (Visser and Both 2005; Primack et al. 2009). Not all species respond to climate change in a similar way; species interactions are therefore also changing (Primack et al. 2009) and likely further facilitating the development of novelty in ecosystem functioning.

In Section 14.1 we illustrated the changes in species interactions as a result of climate change with the fish assemblages in the northeast Atlantic. Climate change has also led to a mismatch in the temporal and/or spatial occurrence of some terrestrial herbivores and food plants (Schweiger et al. 2010) and predators and prey (Visser and Both 2005; Both et al. 2009). Temporal and spatial incongruities are also expected for parasites and their hosts (Thomson et al. 2010). Mismatches are perhaps especially detrimental for mutualistic interactions, including pollination. In addition to a lack of overlap in spatial and temporal space between plants and pollinators due to climate change (Hegland et al. 2009), mismatches in pollinator body size distribution, nectar quantity and quality, pollinator energy demand, plant community structure and pollinator community structure have been observed or are hypothesized as likely to occur due to climate change (reviewed in Schweiger et al. 2010). Other kinds of interactions likely also have subtle but significant mismatches that may contribute to ecosystem novelty (see Chapter 10

for more discussion on this topic; also see Chapters 15 and 40 for case studies).

14.4.4 Land-use practices

Human populations and their use of land is the largest onsite influence responsible for the creation of novel ecosystems in terrestrial systems. As noted earlier (Section 14.2), predicting the effects of human land use on the biosphere is difficult because human interactions with terrestrial ecosystems are exceedingly complex and dynamic. Humans alter terrestrial ecosystems both intentionally and unintentionally and these alterations depend on a variety of interactions including human population density, technical capacity, mode of resource use and the use of opportunities afforded by native and transformed ecosystems. All of these factors interact and are influenced by the considerable natural variation in terrestrial ecosystem form and process (Ellis 2011). Ellis (2011) has analyzed this effect in detail and concluded human populations and their use of land have transformed most of the terrestrial biosphere into anthropogenic biomes (anthromes), causing a variety of novel ecological patterns and processes.

A good example is the transformation that has occurred on islands in the Caribbean such as Puerto Rico, which is detailed in Lugo et al. (2012; see also Chapter 9). The 8959 km^2 island was completely forested when Europeans arrived, despite earlier occupation by indigenous people. Puerto Rico's forest cover fell from almost 100% in 1493 to 6% by the late 1940s before recovering to 57% by 2003 due to agricultural abandonment (Brandeis et al. 2007).

The vegetation that emerged from Puerto Rico's abandoned agricultural lands, which in 1966 covered 68.5% of the island, was different from native plant communities present before deforestation (Lugo and Helmer 2004). These plant communities differed in the diversity of introduced species, which have mixed with natives to form novel plant communities. Despite this new mix of species, after several decades these novel forests developed a physiognomy and structure similar to that of mature native forests. Did the island fauna exhibit similar patterns to the flora? According to Lugo et al. (2012) large numbers of introduced faunal species occur throughout the Caribbean, many of which are naturalized and even locally adapted to the novel plant communities that now dominate the islands. Because relatively few of the native and endemic species have become extinct, they suggest that integration of native animals into novel plant and animal communities is possible.

14.4.5 Combination of drivers

With the complexity of human influence, it is not uncommon that novelty can arise from a combination of different human interventions. For example, the range expansion and population eruption of the native mountain pine beetle (*Dendroctonus ponderosae*) that has resulted in the death of millions of hectares of lodgepole pine (*Pinus contorta latifolia*) in central British Columbia is blamed on two anthropogenic factors: warmer winter temperatures that do not kill overwintering larvae and fire suppression (Cudmore et al. 2010; Klingenberg et al. 2010; Safranyik et al. 2010). In turn, the extirpation of these forests is leading to changes in carbon (Pfeifer et al. 2011) and nitrogen cycling (Griffin et al. 2011), lowered probability of crown fire (Simard et al. 2011) and increased mortality in replanted stands due to migration of the Warren root collar weevil (*Hylobius warreni*; Klingenberg et al. 2010).

In Puerto Rico, the expansion of pastures dominated by introduced African grasses and urban grassy areas have not only introduced a new fire disturbance regime to the island, but have also influenced the success and naturalization of introduced granivorous birds. In a forested island with only 2 native granivorous bird species, today the island has 18 granivorous bird species all associated with novel grass communities (Raffaele 1989).

The introduction of novel herbivores, predators and mutualists can all result in the creation of novel ecosystems by changing species composition and interactions, including among plants. As described earlier, invasion of Christmas Island by yellow crazy ants and the range expansion by the mountain pine beetle have both created novel plant assemblages. A change in the pollinator community can likewise result in the creation of novel plant assemblages. The introduction of mammalian predators to New Zealand resulted in the depletion of the endemic avifauna and significantly reduced recruitment of an endemic forest understory shrub (*Rhabdothamnus solandri*) that relied on the native birds for pollination (Anderson et al. 2011). Changes such as these are likely to be slow and difficult

to detect, but are nonetheless important drivers in the formation of novel ecosystems.

14.5 WHAT CAN NOVEL ECOSYSTEMS DO?

As the amount and rate of ecosystem alteration increases globally, an increasing proportion of the world's ecosystems move into novel configurations and compositions. The amount of primary habitat available for native species is therefore declining, especially in rapidly changing situations such as urban areas. An increasing number of native fauna, including rare, threatened and endangered species, rely on these novel ecosystems to meet their resource requirements and/or provide landscape connectivity for dispersal (Jones and Bock 2005; Vander Zanden et al. 2006; Morrison 2009). The use of novel ecosystems by native fauna primarily occurs when: (1) these systems provide additional resources to fauna that ameliorate effects of change; (2) the novel system provides structure and resources similar to the native habitat; and/or (3) the native habitat has biotic interactions limiting its suitability.

14.5.1 Help mitigate effects of climate change

As noted previously, there is mounting evidence that climatic changes are having direct effects on animal populations through weather-related effects on vital rates, abundances and distributions. In birds, climate-induced phenological shifts in migration timing (Butler 2003; Murphy-Klassen et al. 2005; MacMynowski et al. 2007) and initiation of breeding (Crick and Sparks 1999; Dunn and Winkler 1999) which influence survival and fecundity, respectively, have been documented. As noted by Seavey et al. (2008), effects of climate change on bird populations may be mediated indirectly through changes in food availability or effects on competitors and predators. These climatic changes may reduce the quality of existing habitat to pre-disturbance species. In these situations, novel ecosystems with a mixture of native and non-native species may provide more resources than what was available to these birds under earlier climate regimes.

An excellent example is the recent changes in blue and great tit (*Cyanistes caeruleus* and *Parus major*) breeding habitat in the UK (summarized in Hobbs et al. 2009). The two titmice species are common nesting species in oak (*Quercus* spp) woodlands throughout the British Isles. Recent evidence demonstrates both species are laying eggs earlier in the season in response to climate cues, but there is no evidence that food resources used by titmice during the breeding season (larval and adult insects) are also emerging earlier. A recent inhabitant of these woodlands, the Turkey oak (*Quercus cerris*), provides the food resources available during the early part of the breeding season. This species was native to the British Isles prior to the last glaciations (~130,000–115,000 years BP), and was reintroduced by humans in the past 300 years. There is no evidence the species is invasive, although there have been concerns raised in conservation circles. Native insects preferred by the titmice use the Turkey oak as a host. However, more importantly, the Turkey oak is one of the few hosts for insects migrating northward in the early spring before the normal food sources of the titmice emerge. In this novel ecosystem, the 'non-native' Turkey oak potentially provides additional resources for the titmice under these changed abiotic conditions (assuming the non-native insects do not negatively affect the availability of the native species later in the season).

14.5.2 Mitigate effects of land-use changes

Most human land-use transformation results in land that is managed intensively for specific production or is built over, and these habitats are not considered novel ecosystems (see definition in Chapter 6). However, small areas of novel ecosystems may occur in pockets at the margins or interstices of productive or built environments; novel species interactions may therefore occur in intensively managed systems. Using agricultural lands and managed plantations, we illustrate the importance of these novel ecosystems to faunal conservation. The importance of novel ecosystems in human settlements is discussed in more detail in Chapter 38.

Approximately 37% of the globally available land area is currently in agricultural production and world food demand is predicted to double by 2050 (Tilman et al. 2001; Green et al. 2005). The need for increased food production has resulted in agricultural intensification and the resulting land-use change is currently the greatest extinction threat to fauna (Donald et al. 2001; Foley et al. 2005; Green et al. 2005; Millennium

Ecosystem Assessment 2005). Cleared landscapes support few species and these species are often habitat generalists and tolerant of humans (Green and Catterall 1998; Hobbs et al. 2003). In these agricultural areas, little remains of pre-settlement conditions and the majority of faunal habitat is provided by novel ecosystems that resulted from these extensive landscape transformations. In most situations, complete restoration of pre-agricultural habitat is likely to be prohibitively expensive on a large spatial scale.

The ecological and economic need to restore faunal habitat has resulted in governments channeling resources into mitigating the effect of intensive agriculture by creating novel ecosystems through voluntary agri-environment programs that reimburse farmers to undertake practices aimed at benefiting fauna, e.g. set-asides that leave field margins uncropped (United States Department of Agriculture 2003; Benton 2007). Agri-environment schemes designed to promote these novel ecosystems have been a major policy instrument for the past two decades, and more than 100 programs that aim specifically at developing more conservation value of agro-ecosystems exist in the member countries of the Organization for Economic Co-operation and Development (Aviron et al. 2009). In Europe and North America, nearly $5.25 billion has been spent annually on agri-environment programs (Donald and Evans 2006; Benton 2007).

The majority of these agri-environment schemes focus on programs that can be implemented on individual farms such as the USDA's Conservation Reserve Program (CRP) and affiliated USDA–state-agency partnerships. These programs provide financial and technical assistance to landowners and managers to remove highly erodible lands from agricultural production and establish permanent cover to achieve natural resource conservation objectives, including improved faunal habitat (Brennan and Kuvlesky 2005; Haufler 2005; Gill et al. 2006). Several dozen conservation practices have been identified under the CRP and related programs that result in diverse flowering plants and sequential bloom throughout the growing season and/or improved nest sites for native bees (USDA 2008). Incorporating patches of semi-natural vegetation such as hedgerows into the agricultural landscape has been demonstrated to benefit native pollinators, including those that are otherwise uncommon in the landscape (Hannon and Sisk 2009).

Although the CRP has been in existence for several decades, few studies experimentally document the success of these programs in restoring wildlife habitat. One exception is a study by Gill et al. (2006) who revegetated 92.4 ha of agricultural lands in northeastern Maryland which were historically coastal, open savannah-grasslands that supported the extinct heath hen (*Tympanuchus cupido cupido*). In 1999 they planted 10 native grass species augmented with "small bags of assorted prairie flowers". Five years after planting they documented a cumulative plant species list of 261 species, 105 of which were introduced. The authors predict cumulative plant species richness will stabilize at about 300 species; the ratio of introduced to native is not clear but will fluctuate depending on management and climatic conditions. The revegetation response suggests a seemingly inexhaustible bank of dormant but viable seeds in the soil, remarkable given the more than 200-year history of intensive pastoral and agricultural use of this area (the most recent of which consisted of row-crop monocultures). However, this system is a novel ecosystem that will always contain a mixture of introduced and native species like most CRP sites.

In terms of fauna response, Gill et al. (2006) monitored grassland birds, taxa of international conservation concern. The bird response to the revegetation was equally impressive. Although these sites did not support the full complement of extant grassland species, this novel ecosystem is providing nesting habitat for several grassland-obligate species of conservation concern. For example, the grasshopper sparrow (*Ammodramus savannarum*) and dickcissel (*Spiza americana*) colonized the restored grasslands in the first year and established sustainable breeding populations within 5 years. Whether or not these novel ecosystems are high-quality habitat is yet to be determined, but the return rates and reproductive success data are suggestive of suitability.

There is an extensive literature on the biodiversity value of plantations (e.g. Lindenmayer and Hobbs 2004; Brockerhoff et al. 2008). For example, since WWII extensive acreages of Eucalyptus woodlands were cleared in Australia for croplands and livestock range. Concurrently, the increasing shortfall projected in timber production emanating from the decline in available domestic native forest resources has provided the commercial stimulus for trebling the rate of farm forestry plantations (to 80,000 ha annually) by 2020. Although forest plantations generally do not support the biodiversity of remnant woodlands, globally they do provide habitat for some species, including some

of conservation concern (Newton 1996; Hobbs et al. 2003; Moser and Hilpp 2003).

Lindenmeyer et al. (2008) document the development of a novel assemblage of bird species in southeastern Australia woodlands over a 7-year period in response to recently established introduced pine plantations. Their data suggest these novel ecosystems, which are dominated by an introduced species (radiata pine, *Pinus radiata*), are providing habitat for regionally declining species such as the rufous whistler (*Pachycephala rufiventris*) which likely declined when the original open-woodland *Eucalyptus* were cleared for livestock grazing. The presence of this novel ecosystem also appears to have facilitated the westward expansion of native bird species such as the superb lyrebird (*Menura novaehollandiae*) and the olive whistler (*Pachycephala olivacea*) into nationally endangered open-woodland vegetation types where they have not previously occurred. The effects of these range expansions on resident woodland birds are unknown.

14.5.3 Provide habitat and food resources required by native fauna

Novel ecosystems can provide structural and or nutritional resources for native species. The potential role of novel ecosystems in providing resources for rare native species is likely to be particularly important in situations when restoration of the native species that formerly provided shelter or an energy source is impractical due to limited economic resources or changes in the physical environment (Schlaepfer et al. 2011). For example, Botteri's sparrow (*Aimophila botteri*) is a bird of tall grasslands that disappeared from Arizona following a period of extreme drought and heavy livestock grazing in the 1890s. Botteri's sparrow returned to Arizona in the 1970s but occurs primarily in grasslands dominated by non-native lovegrasses (*Eragrostis* spp); native grasslands with tall structure are restricted to <5% of their former range (Jones and Bock 2005). Jones and Bock (2005) studied the abundance and reproduction of this species across the state and concluded that the novel grasslands were not an ecological trap but provide the tall grass structure required by this species for nesting and survival. Novel ecosystems providing structural resources have been observed in other vertebrates where structure may be a more useful predictor of habitat than floristics. Examples include the: (1) southern emu-wren (*Stipiturus malachurus*; Wilson and Patton 2004; Maguire 2006); (2) many North American breeding grassland and riparian breeding birds (Gill et al. 2006; Bajema et al. 2009; Kennedy et al. 2009; Fisher and Davis 2010; Paxton et al. 2011); (3) Sarus crane (*Grus antigone*; Sundar 2009); and (4) several marsupials including the red-cheeked dunnart (*Sminthopsis virginiae*; Braithwaite and Lonsdale 1987) and long-nosed bandicoot (*Perameles nasuta*; Chambers and Dickman 2002).

In addition to structural resources, non-native fruit quantity has recently been correlated with abundance of native avian frugivores (Davis 2011; Gleditsch and Carlo 2011) and non-native prey enhances the abundance of native predators (Schlaepfer et al. 2011). In some cases, non-native plants increase the overall diversity and availability of resources for native bees (Winfree et al. 2007; but see Williams et al. 2011).

14.5.4 Mitigate effects of limiting biotic interactions in native habitat

Although to our knowledge this has not been documented, it is possible that novel ecosystems could be refugia from native habitat that has biotic interactions that limits the native species persistence. These biotic interactions could be predation and/or competition and the novel ecosystem could provide shelter from competition and/or predation. In the previous section we discussed the preference of some species, e.g. long-nosed bandicoot, Botteri's sparrow, for the structural characteristics of novel ecosystems. It is not clear if those preferences are: (1) a result of choice from a limited range of alternatives and the novel ecosystem provides the structure that most closely approximates the native habitat; or (2) the novel ecosystem provides a structural environment that reduces the risk of predation and/or competition as compared to the native habitat. Future investigations on faunal use of novel ecosystems should be designed to separate these alternative scenarios.

14.6 UTILIZING NOVEL ECOSYSTEMS TO ACHIEVE FAUNAL CONSERVATION GOALS

Today's managers must recognize that many natural systems of the past are changing forever towards a

new natural state thanks to drivers such as climate change, increased urbanization, introduced species and other transformative land-use/land-cover changes that are not reversible with restoration. We agree with Bridgewater et al. (2011) that such ecosystems are a blind spot in global conservation and restoration priorities as well as in discussions of environmental management and sustainable development. We argue that novel ecosystems are now critical for maintenance of faunal diversity at the genetic, species and ecosystem levels, and restoration goals to eliminate novelty might not always benefit faunal conservation. In some cases, novel ecosystems may provide the only habitat available for species of conservation concern. In other cases, novel ecosystems may provide habitat value not provided by native habitats that have evolved under different climatic and landscape conditions. The challenge to managers is to recognize when to intervene and when to roll with the punches in relation to the biotic composition of novel systems. The large cost of reversing the effects of human activities is an incentive to minimize them to the extent possible and, where not possible, to use scientific understanding to recognize when a novel assemblage is persistent and desirable. Chapters 3 and 18 provide some insights and analytical frameworks for assisting with these difficult decisions.

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NOVEL ECOSYSTEMS

INTERVENING IN THE NEW ECOLOGICAL WORLD ORDER

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