Editorial

Greener pastures – but at what cost?

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In this, my last article as Editor of *The Weta* prior to moving on to (metaphorically) 'greener pastures' in Western Australia, it seems only appropriate to consider the implications of that age-old adage, 'the grass is always greener'. Over the past 10 years, New Zealand, like much of the rest of the world, has been under-going the most significant period of land transformation since the post-WW2 re-settlement and re-building of the economy. Those beautiful green pastures we see popping up in the most unexpected places, including Canterbury and the Mackenzie country, may indeed produce valuable export dollars that make the country wealthy, but they also come at a significant cost to the environment. Overseas, it is increasingly recognised that intensification of agriculture is associated with a whole suite of negative effects on the environment that are so severe they have been referred to colloquially as the 'second Silent Spring' (Krebs *et al.* 1999). Yet, the potential costs of land-use intensification in New Zealand have been surprisingly little remarked upon, particularly with regard to native invertebrate communities. I would argue that we, the New Zealand public, are simply subsidising the profits that companies make from exporting the products of high-intensity land use, by internalising the costs of environmental degradation within the country. One way or another, these environmental costs will ultimately be paid through loss of biodiversity, increased costs of environmental restoration, or the tarnishing of the tourist industry's 'clean, green' PR slogan. Over the next three years, the principal connection I will retain in New Zealand will be with several research projects investigating the ecological dimension to the impacts of land-use intensification on natural ecosystems within production landscapes.

In the U.K. and Europe, the 'smoking gun' for agricultural intensification has been the dramatic decline in farmland birds in recent decades, including such 'indestructible' pest species as sparrows and starlings, due to direct loss of food resources (seeds and invertebrate prey) and reduction in habitat heterogeneity (Krebs *et al.* 1999, Donald *et al.* 2001, Benton *et al.* 2003). These impacts have been explained mostly as a direct consequence of competition for environmental resources between humans and other species. For example, Krebs *et al.* (1999) consider that "in general terms, [land-use] intensification is about making as great a proportion of primary production as possible available for human consumption", and this mirrors the standard definition of agricultural intensity in terms of production output per unit area per unit time (Turner & Doolittle 1978). However, I would argue that the ecological impact of agriculture is mainly about the indirect effects of the *inputs* required to increase the intrinsic primary productivity of the site (i.e. the removal of site limitations of primary productivity) by external subsidisation of water (irrigation) and nutrients (fertiliser), and removal of competitors (cultivation, herbicides) and consumers (insecticides), and then only secondarily about the direct removal of resources by maximising *outputs* of harvest and consumption by humans. Agricultural inputs typically alter soil geochemistry and the physical characteristics of the site to make it less suitable as habitat for native organisms, and this has been shown to have long-term (perhaps permanent) consequences for biodiversity on abandoned farmland, even when outputs for human consumption cease (Cramer *et al.* 2008). In addition, most of the major mechanisms of effect are likely to scale directly with inputs rather than outputs, not least of which is because many detrimental impacts are due to inefficiencies in the usage or targeting of the inputs towards production. For example, it is well known that the higher the single-pulse application of nitrogen fertiliser, the greater the loss rate of nitrogen to adjacent systems due to volatilisation or run-off prior to local uptake by the crop.

The farmland bird declines in Europe have stimulated a wealth of studies focusing on the impact of land-use intensification on other taxa, such as plants and invertebrates. For these groups there has been similar evidence for widespread declines in the populations of many species due to agricultural intensification (Benton *et al.* 2003, Conrad *et al.* 2006, Marini *et al.* 2009), with significant effects in some cases on the provision of ecosystem services such as crop pollination (Kremen *et al.* 2002). The range of mechanisms underlying these effects has also been highlighted in studies showing the potential to reverse at least some ecological impacts at a local scale, through reductions in livestock density (e.g., Littlewood *et al.* 2008) and fertiliser inputs (e.g., Dennis *et al.* 2004), amongst other factors linked to land-use intensity.

What is surprising, is that despite ample evidence that biodiversity and related services do indeed decline with agricultural intensification, there has been no critical assessment of the actual shape of the relationship between biodiversity and land-use intensity, until a new study published just last month by David Kleijn and colleagues (Kleijn *et al.* 2009). Establishing the nature of the biodiversity versus land-use change relationship unquestionably represents the central foundation on which conservation management in agro-ecosystems should be based, and arguably it is a sad indictment on the state of the field that such a relationship has only just now been derived. After all, there would be fundamentally different implications if biodiversity loss scaled linearly with land-use intensification, in which case a given unit of conservation management effort would be just as effective at all levels of intensification, compared to a situation where biodiversity loss scaled exponentially, or as a threshold function, of intensification, in which case conservation management actions might well be less effective at high land-use intensity.

The novelty of their approach was that Kleijn *et al.* (2009) linked a biodiversity estimate (for plants, unfortunately, not invertebrates) to nitrogen fertiliser inputs as a management indicator of land-use intensification in a large number of agricultural fields across Europe, thereby covering a wide range from very extensive to extremely intensive land use. After accounting for regional differences in environmental conditions, Kleijn *et al.* (2009) found that plant species richness declined in a negative exponential relationship (not linearly) with increasing land-use intensity. This suggests that the effects of intensification were most severe per unit land-use change in extensively-managed, species-rich agricultural areas, while above a 'threshold' of around 100 kg N addition per hectare per year, plant species richness declined to levels where further intensification had little or no discernible effect (although the apparent 'threshold' was not statistically tested, *per se*).

The implications of these findings are clear. Dollar-for-dollar, conservation initiatives will be most effective if they are preferentially implemented in extensively-managed farming systems that still support high levels of biodiversity. By contrast, in high-intensity farming systems, the economic cost of reducing intensity is likely to be far greater due to the higher investment in farm infrastructure, while the biodiversity gain per unit reduction in intensity will be far smaller. Consequently, the cost-benefit ratio for conservation management is likely to be orders of magnitude in favour of applying conservation actions in low-intensity farming regions, and this is only going to widen the gulf that already exists between land allocated to intensive agricultural production versus land set-aside for nature conservation. Furthermore, recent studies suggest that there is only relatively weak evidence for net biodiversity benefits associated with this 'land-sparing' rationale to land-use intensification (Ewers *et al.* 2009), and it runs counter to the growing recognition of the importance of maintaining biodiversity within production landscapes (Norton 2000).

Whether these findings can be extrapolated to other taxa and other situations remains unclear. Certainly, at some sites where Kleijn and colleagues had previously sampled specific invertebrate groups, including bees, grasshoppers and spiders (Kleijn *et al.* 2006), there was associational evidence that plant species richness was indeed positively correlated with invertebrate species richness, and therefore that the exponential relationship with land-use intensification might well hold across taxa. Across different regions of the world, similar trends in land-use intensification are been played out in both temperate and tropical biomes, and there is no reason to suspect that the impacts on biodiversity will not hold more widely also. If anything, it is even possible that the effects of intensification will be more severe in other regions than observed by Kleijn *et al.* (2009) in Europe. They make the interesting point that their sites in Europe have been exposed to agriculture for a much longer period than in most other regions of the world, and therefore that a significant proportion of the remaining flora and fauna may well be fairly resilient to the adverse effects of land-use intensity. Consequently, the effects of agricultural intensification on biodiversity may be even more pronounced outside Europe, in countries such as New Zealand, with a recent history of agricultural development (McLeod & Moller 2006).

Finally, it is worth pointing out that the relationship observed in Kleijn *et al.* (2009) was only derived for the impacts of land-use intensification on biodiversity within production systems themselves, and it is not yet known what the potential shape of the relationship will be between land-use intensification and biodiversity loss in adjacent nature reserves or semi-natural habitats embedded within production landscapes. It is well recognised that agricultural inputs do not stay put on farms, but have a wide range of impacts on adjacent systems, most notably where increased nutrient run-off affects aquatic insect communities in adjacent waterways. However, this has not translated into a widespread understanding of what the impacts of intensification will be on terrestrial invertebrate communities in the forest ecosystems that formerly dominated high-production farming landscapes of New Zealand. This represents a crucial missing piece of the puzzle in terms of understanding the trade-offs to be made between agri-environment schemes that aim to reduce land-use intensification, and land-sparing arguments for the localised increase in land-use intensification in some areas to offset preservation of 'wild nature' in other areas. If high-intensity farming has significant off-site effects on biodiversity on land spared for nature, then this would seriously undermine land-sparing arguments.

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