LAND USE IN LCA

Land use impacts on biodiversity from kiwifruit production in New Zealand assessed with global and national datasets

Carla R. V. Coelho · Ottar Michelsen

Abstract

Purpose Habitat loss is a significant cause of biodiversity loss, but while its importance is widely recognized, there is no generally accepted method on how to include impacts on biodiversity from land use and land use changes in cycle assessment (LCA), and existing methods are suffering from data gaps. This paper proposes a methodology for assessing the impact of land use on biodiversity using ecological structures as opposed to information on number of species.
Methods Two forms of the model (global and local scales) were used to assess environmental quality, combining ecosystem scarcity, vulnerability, and conditions for maintaining biodiversity. A case study for New Zealand kiwifruit production is presented. As part of the sensitivity analysis, model parameters (area and vulnerability) were altered and New Zealand datasets were also used.
Results and discussion When the biodiversity assessment was implemented using a global dataset, the importance of productivity values was shown to depend on the area the results were normalized against. While the area parameter played an important role in the results, the proposed alternative vulnerability scale had little influence on the final outcome.
Conclusions Overall, the paper successfully implements a model to assess biodiversity impacts in LCA using easily accessible, free-of-charge data and software. Comparing the model using global vs. national datasets showed that there is a potential loss of regional significance when using the generalized model with the global dataset. However, as a guide to assessing biodiversity impact, the model allows for consistent comparison of product systems on an international basis.

Keywords Biodiversity · Ecoregions · Impact assessment · Kiwifruit · Land use · Scale dependency

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1 Introduction

In the drive for sustainability, knowledge of the environmental impact of products and production systems is essential and life cycle assessment (LCA) is the prevailing framework for analyzing these impacts. Land use and land use changes, in particular extensive changes in agriculture or forestry, are also recognized as impacts that should be included in LCA but, despite being discussed since the early 1990s (Bare 2010), no agreed method for this exists (Milà i Canals et al. 2007). The relationship between land use and impacts on biodiversity has been long recognized (Barnes 1996) and loss of biodiversity is seen as one of the most severe threats to sustainability (Diaz and Cabido 2001). Despite recent focus due to the Convention on Biological Diversity (UNEP 1992), there are few signs of reductions in the rate of loss of biodiversity (Butchart et al. 2010) and changes in land use are often the most important driver (Barnes 1996; Chapin et al. 2000; Sala et al. 2000; Henry et al. 2008; Haines-Young 2009; Lenzen et al. 2009). However, it is not only major changes in land use that are important; small modifications to

land use that are not generally noted when land use changes are assessed, which might be equally important for biodiversity transformation (Barnes 1996; Haines-Young 2009). Barnes (1996) warns of the trap of putting higher value on things that exhibit immediate or obvious merit or which are easily measurable. He argued that a focus on rich ecological systems (e.g., tropical forests) is "inevitably a reactionary rather than proactive course of action" (p. 241) and that focusing on threatened hotspots can be ineffective.

Assessment of land use using LCA is typically done as a product of the area affected, duration of the impact, and a qualitative measure of changes due to the impact (cf. Lindeijer 2000; Milà i Canals et al. 2007). Most proposals have used changes in species richness as a general measure for changes to biodiversity. The use of vascular plants as indicator has in particular been common in LCA (e.g., Muller-Wenk 1998; Köllner 2000; Weidema and Lindeijer 2001; Toffoletto et al. 2007; Schmidt 2008; De Schryver et al. 2010), but other taxonomic groups are used as well (e.g., Geyer et al. 2010; de Baan et al. 2013). Use of selected taxonomic groups to represent biodiversity is debatable, partly because changes in species diversity only capture parts of the concept of biodiversity (cf. Millennium Ecosystem Assessment 2005) and partly because species richness in one taxonomic group has variable correlation with that of other taxonomic groups (e.g., Prendergast and Eversham 1997; van Jaarsveld et al. 1998; Lawton et al. 1998; Similä et al. 2006). Some studies have shown correlations among taxonomic groups (e.g., Koellner 2002; Kati et al. 2004), but it is questionable whether these results are valid outside the area where the background datasets were collected. Interestingly, Gottfried et al. (2012) have documented that an early response to a long-term negative impact on biodiversity might in fact be a short-term increase in species richness, further stressing the fact that assessing changes in species richness can be an inadequate indicator for impacts on biodiversity since an increase cannot necessarily be interpreted as reduced impact. Partly in response to the shortcomings of using species richness as a surrogate for biodiversity, some authors have proposed focusing on other features known to be important for biodiversity, such as water stress (Milà i Canals et al. 2009, 2010), structural features (Michelsen 2008) or a framework based on expert opinions (Penman et al. 2010). Another problem with the proposed methods is the resolution. Some methods are specific to their case study areas and cannot be used elsewhere without intensive data gathering (e.g., Michelsen 2008; Geyer et al. 2010), some are assuming the same impact irrespective of where the land use occurs (e.g., Goedkoop et al. 2009), and yet others have a biogeographic subdivision, but on the scale of biomes (e.g., de Baan et al. 2013). Curran et al. (2011) pointed out that most approaches to biodiversity assessment in LCA are developed for specific regions, and criticized the global

approach at biome level as being too coarse, but even at this scale, de Baan et al. (2013) experienced lack of data with a species approach.

The main purpose of this paper is to present a model that enables assessments of land use impacts on biodiversity in life cycle impact assessment (LCIA), independent of any particular biogeographic region, based on globally available data with a higher resolution than biomes. The proposed method is exemplified with a case study for kiwifruit production in New Zealand. We also performed a sensitivity check for some of the most important assumptions in the model in order to test the strength of the method.

2 Background

Land cover can be classified at different scales. One classification divides the world into 867 distinct ecoregions, which are defined as relatively large units of land containing different natural communities and species, with boundaries that approximate the original extent of natural communities before major land-use change (Olson et al. 2001). An interactive map of these ecoregions, as well as an indicator for the state of conservation for each, can be found on the World Wildlife Fund "WWF WildFinder" website (WWF 2012). Nine ecoregions can be found in New Zealand's North and South islands (Fig. 1), all of them endemic to New Zealand. Ecoregions are the basis for our generic model, following Michelsen (2008).

2.1 Land use in New Zealand

New Zealand also has a national classification system, the Land Environments of New Zealand (LENZ). LENZ uses a combination of climatic and physiographic (landform and soil data) variables to classify New Zealand into areas containing ecosystems of similar type. LENZ is constructed hierarchically with four different resolutions, levels 1–4 dividing New Zealand into 20, 100, 200, and 500 different environments, respectively (MfE 2009). The "Threatened Ecosystem Classification" system provides information on vulnerability, including the amount of remaining indigenous cover and protected areas for all 500 environments included in LENZ (Landcare Research 2012). We use level 4 of LENZ when we use our model specifically to assess kiwifruit production.

In New Zealand, an estimated 50 % of land cover is indigenous vegetation, nearly 40 % is pasture, with the balance being exotic vegetation (9 %) and urban and built-up areas (0.8 %); only 1.5 % of land cover in New Zealand is horticulture (MfE 2007). Although large areas of New Zealand are protected for conservation purposes, Walker et al. (2005) claim that protection is skewed by a tendency

Fig. 1 Ecoregions of New
Zealand and their state of
conservation, based on
information from the World
Wildlife Fund's "WWF
WildFinder" (WWF 2012)
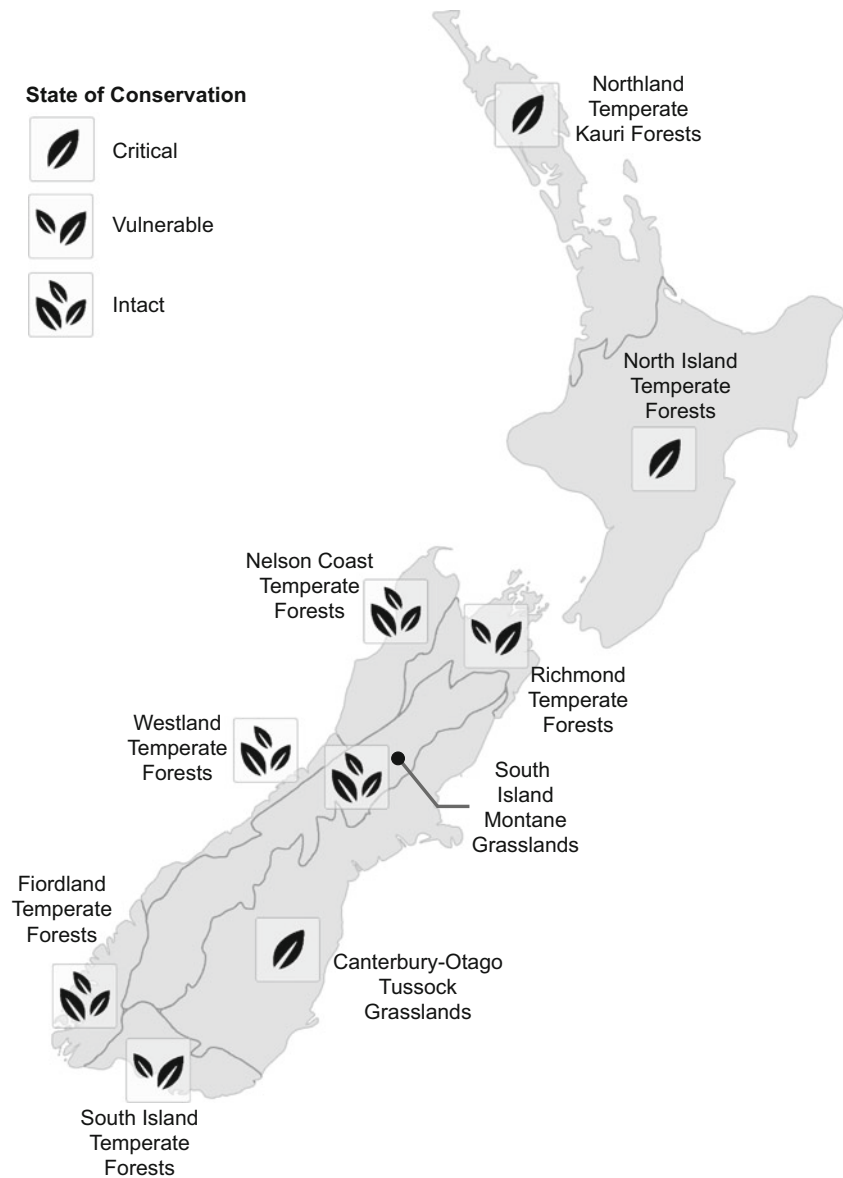
to protect more wet mountainous environments while habitats and ecosystems in productive lowland and montane environments are less protected. As a consequence, indigenous vegetation is mostly found in the alpine and upper montane zones and only traces on warmer, lower montane and lowland zones where productive land-use activities are more widespread.

2.2 Kiwifruit production case study

In this paper, we use kiwifruit production in New Zealand to illustrate a method for assessing impacts from land use on biodiversity. Our focus here is limited to impacts on biodiversity caused by land use and we do not include impacts from land use changes. Because the land use impacts of an agricultural product's life cycle are expected to be dominated

by the cultivation phase, land use impacts from other life cycle stages are not included in this study.

New Zealand is the second largest producer of kiwifruit in the world (Kilgour et al. 2008; FAO 2012). In 2010, the New Zealand total production was 378,500 t, constituting 28 % of world production and a harvested area of ~12,800 ha (FAO 2012). New Zealand's kiwifruit industry has a high focus on environmental performance including efforts in carbon management, use of water resources, lean manufacturing, packaging, transportation and carbon footprint assessment (Peltzer et al. 2011; ZESPRI 2010, 2012).

In order to cover most of the areas used for kiwifruit production in New Zealand, 10 major kiwifruit growing regions are included in this study (Auckland, Gisborne, Hawke's Bay, Katikati, Nelson, Northland, Tauranga, Te Puke, Waikato, Whakatane). These orchards are situated in

only three ecoregions: Northland temperate kauri forests; North Island temperate forests; and Richmond temperate forests (Fig. 1). In total, kiwifruit production in the selected regions occupies 8,900 ha.

As a starting point, we base our model on a method proposed by Michelsen (2008) for boreal forest, adapting it for more generic use by including aspects of a method proposed by Brentrup et al. (2002). For our kiwifruit case study, the functional unit is 1,000 tray equivalents (TE, 1 tray equivalent corresponds to 3.6 kg of green kiwifruit at gate), using data for the production year 2008/09 as an example.

3 Methods

3.1 Method description

According to Milà i Canals et al. (2007), three aspects must be assessed when quantifying land use impacts on biodiversity: the area affected must be identified, the duration of the impact must be assessed, and a quantitative measure of impact on biodiversity must be established and assessed.

Michelsen (2008) suggested an indirect approach for assessment of biodiversity quality (Q) and changes in quality at a given location based on ecosystem scarcity (ES), ecosystem vulnerability (EV), and conditions for maintained biodiversity (CMB), using the formula

$$Q = \text{ES} \times \text{EV} \times \text{CMB}. \tag{1}$$

"Ecosystem scarcity," originally proposed for LCIA by Weidema and Lindeijer (2001), is a measure of the intrinsic rareness of an ecosystem. In this paper, we will apply the ES equation proposed by Michelsen (2008);

$$\text{ES} = 1 - \frac{A_{\text{pot}}}{A_{\text{max}}}, \tag{2}$$

where A_{pot} is the potential distribution of the structure in focus and A_{max} is the potential distribution of the most widespread structure at the relevant level. By the term "structure," we mean an ecologically and geographically defined area separated from other areas due to species composition and/or geophysical conditions. The scale-independent-term structure is used since different spatial scales can be used: biomes, ecoregions, etc. Michelsen (2008) suggested the use of ecoregions since these are the globally available data with the highest resolution (Olson et al. 2001; WWF 2012). Note that in this model, the structures with the highest scarcity thus get a value close to 1.

"Ecosystem vulnerability" is a measure of the present condition of the structure. Different approaches are suggested (Peter et al. 1998; Weidema and Lindeijer 2001; Michelsen 2008). In WWF's (2012) WildFinder, these are given on a three-grade ordinal scale. In order to translate this into a globally available quantitative scale, Michelsen (2008) suggested that the value 1.0 be used for ecoregions with a critical conservation status, 0.5 for ecoregions with a vulnerable conservation status, and 0.1 for intact ecoregions.

"CMB" is an index for the actual impact on biodiversity in the affected area, ranging from 1, which represents no impact on biodiversity, to 0, which represents a complete removal of biodiversity. The CMB index is constructed from a suite of indicators known to be important for biodiversity in an ecosystem, and will thus give an indirect measure of the impact on biodiversity (cf. Larsson 2001). Michelsen (2008) constructed the CMB index for boreal forests using indicators such as amount of dead wood and alien species. Such indicators are not directly transferrable to other ecosystems and a challenge with this method is the need to construct ecosystem-specific indexes. To overcome this challenge, we propose the use of hemeroby values (levels of naturalness) as suggested by Brentrup et al. (2002). Hemeroby values measure the human influence on ecosystems by determining the deviation from naturalness as a result of a specific land use (Kowarik 1999; Brentrup et al. 2002; Walter and Stützel 2009). The scale proposed by Brentrup et al. (2002) provides the naturalness degradation potential (NDP) for 39 land-use types. The scale includes most traditional uses of the land and covers most land uses across the world.

Paracchini and Capitani (2011), in their summary of hemeroby, state that the degree of hemeroby is the result of the impact on a particular area and the organisms which inhabit it. The "cultivation of special crops (e.g., fruits vine)" or "extensive arable land use type" both have NDP of 0.7 (cf. Tables 1 and 2 from Brentrup et al. 2002). The 0.7 value is used as a proxy for kiwifruit orchards and when fitted to the scale of 0 for worst to 1 for best case; in this case study, the CMB is 0.3 for all orchards.

The impact of land use is calculated as the change in quality as a result of the ongoing activity. Both Milà i Canals et al. (2007) and Michelsen (2008) consider a continued land use, such as an orchard, as a suppression of a relaxation process, i.e., as a delay in recovery from an impact to ecosystem quality. In other words, the assumed alternative to the assessed land use is that the area recovers to its potential vegetation. The change in quality can then be calculated as

$$\Delta Q = Q_{t\text{pot}} - Q_{t1}, \tag{3}$$

where $Q_{t\text{pot}}$ is the potential quality of the area (defined as ES × EV) and Q_{t1} the quality of the area at the time (t_1) of the activity (calculated as ES × EV × CMB$_{t1}$). As a consequence,

Table 1 Land use impact on biodiversity from kiwifruit plantations following an adjusted version of the method proposed by Michelsen (2008) using hemeroby as CMB values

Location	TE/ha	A_{pot}	ES	ΔQ	ha/y	$\Delta Q \times$ ha $\times$ y	Ecoregion	EV
Nelson	6,509	13,200	0.997	0.349	0.154	0.054	AA0408	0.5 (vulnerable)
Te Puke	8,451	84,400	0.982	0.687	0.118	0.081	AA0405	1 (critical)
Tauranga	8,125	84,400	0.982	0.687	0.123	0.085	AA0405	1 (critical)
Katikati	7,411	84,400	0.982	0.687	0.135	0.093	AA0405	1 (critical)
Auckland	7,409	29,900	0.994	0.695	0.135	0.094	AA0406	1 (critical)
Whakatane	6,040	84,400	0.982	0.687	0.166	0.114	AA0405	1 (critical)
Northland	5,839	29,900	0.994	0.695	0.171	0.119	AA0406	1 (critical)
Waikato	5,770	29,900	0.994	0.695	0.173	0.121	AA0406	1 (critical)
Gisborne	5,173	84,400	0.982	0.687	0.193	0.133	AA0405	1 (critical)
Hawke's Bay	4,868	84,400	0.982	0.687	0.205	0.141	AA0405	1 (critical)

TE kiwifruit tray equivalents, A_{pot} potential distribution, *ES* ecosystem scarcity, *EV* ecosystem vulnerability, *CMB* conditions for maintained biodiversity, *Q* biodiversity quality

$$\Delta Q = Q_{tpot} - Q_{t1} = (ES \times EV) - (ES \times EV \times CMB_{t1})$$
$$= (1 - CMB_{t1})(ES \times EV). \qquad (4)$$

3.2 Sensitivity of the method

This section addresses the sensitivity of our method assumptions, because (1) it is possible to have different perspectives when choosing A_{max}; (2) different spatial resolution can be selected for the structures in focus; and we discuss (3) the choice of values for EV in Michelsen (2008) and consider other values.

3.2.1 Choice of A_{max}

Michelsen (2008) normalized the value of ES using the most widespread ecoregion as A_{max}, therefore using the Sahara Desert which has a distribution of 4,639,900 km^2 (WWF ecoregion PA1327) as A_{max}. Here, we test the use of the most widespread ecoregion in New Zealand, AA0405 North Island temperate forests with a distribution of 84,400 km^2, and the sum of the distribution of all ecoregions in New Zealand, in total 265,000 km^2, as alternatives. When $A_{max} = A_{pot}$, ES will necessarily be 0 and all impacts in such an ecoregion will by definition be 0. To avoid this, A_{max} is set to 90,000 km^2 when AA0405 is used as A_{max}.

3.2.2 Use of national classifications

While ecoregions divide the ecosystems of the world on a global scale, many countries have, at least for country-specific purposes, classifications with a higher resolution. Here, we use the level 4 classification of LENZ as an alternative to ecoregions. Regarding the vulnerability of the environments included in our model, New Zealand also has

Table 2 Land use impact on biodiversity from kiwifruit plantations when A_{max} is changed from using values for ecoregion PA1327 (Sahara Desert) to total area of ecoregions in New Zealand (TotNZ) and the most widespread ecoregion in New Zealand (90,000 km^2)

Location	ES			ΔQ			$\Delta Q \times$ ha $\times$ y		
	A_{max} PA1327	A_{max} TotNZ	A_{max} 90,000 km^2	A_{max} PA1327	A_{max} TotNZ	A_{max} 90,000 km^2	A_{max} PA1327	A_{max} TotNZ	A_{max} 90,000 km^2
Nelson	0.997	0.950	0.853	0.349	0.333	0.299	0.054	0.051	0.046
Te Puke	0.982	0.682	0.062	0.687	0.239	0.022	0.081	0.028	0.003
Tauranga	0.982	0.682	0.062	0.687	0.239	0.022	0.085	0.029	0.003
Katikati	0.982	0.682	0.062	0.687	0.239	0.022	0.093	0.032	0.003
Auckland	0.994	0.887	0.668	0.695	0.311	0.234	0.094	0.042	0.032
Whakatane	0.982	0.682	0.062	0.687	0.239	0.022	0.114	0.040	0.004
Northland	0.994	0.887	0.668	0.695	0.311	0.234	0.119	0.053	0.040
Waikato	0.994	0.887	0.668	0.695	0.311	0.234	0.121	0.054	0.041
Gisborne	0.982	0.682	0.062	0.687	0.239	0.022	0.133	0.046	0.004
Hawke's Bay	0.982	0.682	0.062	0.687	0.239	0.022	0.141	0.049	0.004

ES is ecosystem scarcity, *Q* is biodiversity quality

the Threatened Ecosystem Classification, which is based on the assessment of indigenous cover. To get more accurate data, the Land Cover Database, LCDB2 (MfE 2009) was used in combination with LENZ level 4 to determine in which LENZ environments the orchards occur and how much of the indigenous vegetation is left in those environments. QuantumGIS (QGIS 2012), an open-source geospatial information system software, was used for data processing.

To identify where orchards are located, we used the most up-to-date, publicly available version of land cover data (the second edition of New Zealand's Land Cover Database (LCBD2)) from the Koordinates portal (www.koordinates.com). The polygons of the LCDB2 class "orchards, vineyard, and perennial crops" were used as boundaries for the areas of each LENZ level 4 environment, and the area of each LENZ level 4 environment in the orchards was calculated. This process was carried out for each of the 10 selected kiwifruit growing regions. Following this approach, the ecosystem vulnerability cannot be classified following the ordinal scale Michelsen (2008) originally used. Here, Eq. (6) (below) is used to calculate values for ecosystem vulnerability for each LENZ structure based on amount of indigenous cover left.

LENZ has much higher resolution than the data on the kiwifruit cultivation areas, and more precise geospatial data for the location of orchards were unavailable. Therefore, ES was calculated using Eq. (5), which is a weighted average for each kiwifruit growing region, following

$$\overline{\text{ES}} = \sum_{i=1}^{n} \left(\left(1 - \frac{A_{\text{pot}_i}}{A_{\max}} \right) LU_i \right), \tag{5}$$

where $\overline{\text{ES}}$ is the weighted average of the scarcity of the area in focus (as a kiwifruit growing region), A_{pot_i} is the potential distribution of environment i, $A_{\max}$ is the most widespread environment at the same level (here LENZ level 4), and LU_i is the percentage of environment i for the total area included.

3.2.3 Scaling ecosystem vulnerability

Michelsen (2008) used the state of conservation from WWF's Wildfinder (WWF 2012) for scaling ecosystems' vulnerability. Here, this is adjusted to an exponential function based on knowledge of the fraction of the original vegetation lost (FL). Following the idea from Michelsen (2008), EV=0 when the entire structure is still present (FL=0), and EV=1 when the entire structure is gone (FL=1). In order to construct the function, it was also assumed that when 50 % of the fraction was lost, EV is 0.25, giving the function

$$EV = -0.125 + 0.125 \exp(2.1972 \times FL). \tag{6}$$

The steepness of the curve (Fig. 2) is debatable, and a universal curve probably does not exist (cf. de Schryver et al. 2010). Lindgaard and Henriksen (2011) provide estimates on the relationship between fraction of structures lost and how endangered they are. Based on this information combined with Eq. (6), we suggest that 1.0 is still used for critical ecoregions, but that vulnerable ecoregions are adjusted to 0.45 and intact ecoregions to 0.09. The values based on this approach are thus comparable with the values applied by Michelsen (2008).

As for ecosystem scarcity, ecosystem vulnerability (EV) must also be assessed as a weighted average when more than one structure is included in the area considered, following

$$\overline{\text{EV}} = \sum_{i=1}^{n} (\text{EV}_i \times \text{LU}_i), \tag{7}$$

where $\overline{\text{EV}}$ is the weighted average, EV_i is the ecosystem vulnerability of structure i following Eq. (6), and LU_i is the percentage of the total land use found in structure i.

4 Results

Table 1 shows the results of the land use impact of producing 1,000 TE at the included locations following the proposed default values for ES, EV, and CMB, where hemeroby values are used for CMB. The results are sorted from lowest to highest impact.

Figure 3 is a graphical representation of parameters of the model and of the results. Note that the values are adjusted to fit a 0 to 1 scale for graphical representation.

Table 2 shows the results when $A_{\max}$ is changed to the sum of all ecoregions in New Zealand (TotNZ) and to the
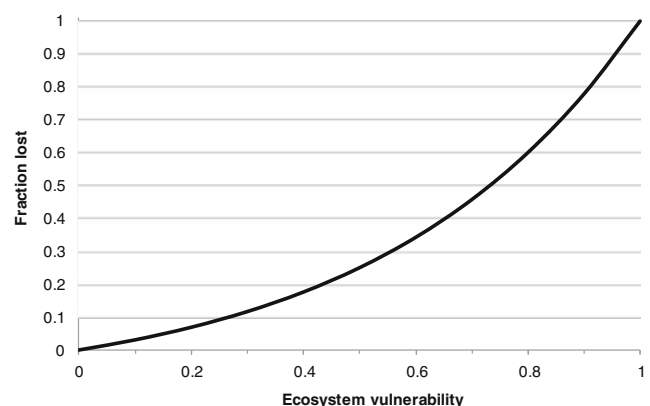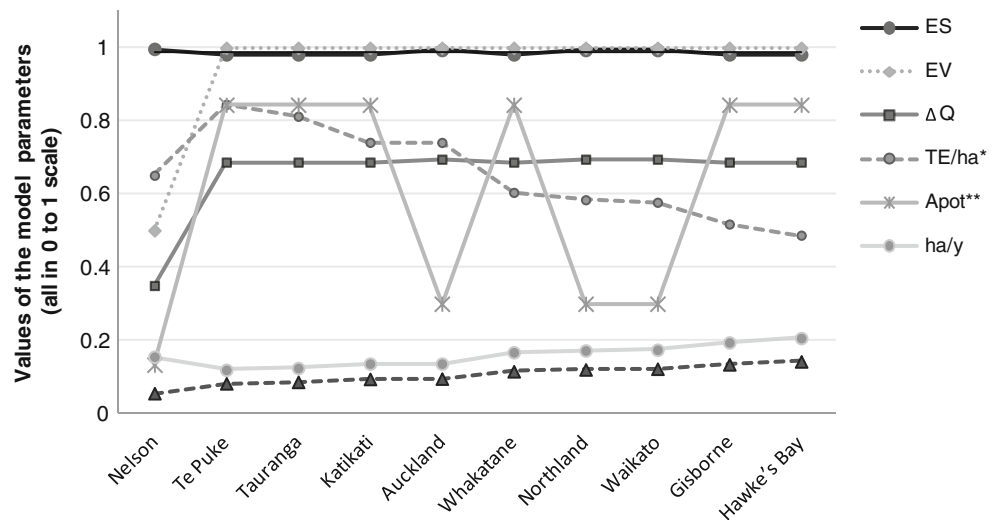

Fig. 2 Graphical presentation of the relationship between fraction lost of a structure and its ecosystem vulnerability, following the proposed equation (Eq. (6))

Fig. 3 Graphical presentation of the model parameters and results shown in Table 1. * for TE/ha, the scores are in 10^{-5}. ** for A_{pot}, the scores are in 10^{-6}

largest New Zealand area (here 90,000 km² is used to avoid A_{max} being equal to A_{pot}).

The nonlinear adjustment of EV (see Section 3.2.3) has only minor implications when used at ecoregion level and for New Zealand kiwifruit production is only relevant for the Nelson region, where EV is reduced from 0.50 to 0.45. When the score for Nelson shown in Table 1 is recalculated with this adjustment, the total score is reduced from 0.054 to 0.048. The choice of A_{max} has a larger impact (Table 2); the absolute values of ES and ΔQ decline with a smaller A_{max} (Table 2). This influences how the different aspects impact on calculating the total impact from land use. As can be seen from Fig. 4, when A_{max} is reduced from the size of the Sahara (4,639,900 km²) to the area of New Zealand (265,000 km²) or the most widespread ecoregion in

New Zealand (90,000 km²), not only are the absolute scores reduced, but the ranking among the kiwifruit growing regions also changes.

In Table 3, the results using LENZ level 4 are shown; the underlying data showing the impact from different structures are given in the Electronic Supplementary Material.

5 Discussion

At present, there is no generally accepted method for how to include impacts from land use and land use changes in LCA. For land use impacts on biodiversity, several of the methods proposed make use of changes in species diversity and in

Fig. 4 Total land use impact on biodiversity for the different locations when A_{max} is altered from the most widespread ecoregion worldwide (Sahara Desert), to the total area of ecoregions in New Zealand or the most widespread ecoregion in New Zealand. For Nelson*, EV is reduced from 0.50 to 0.45 (see text)
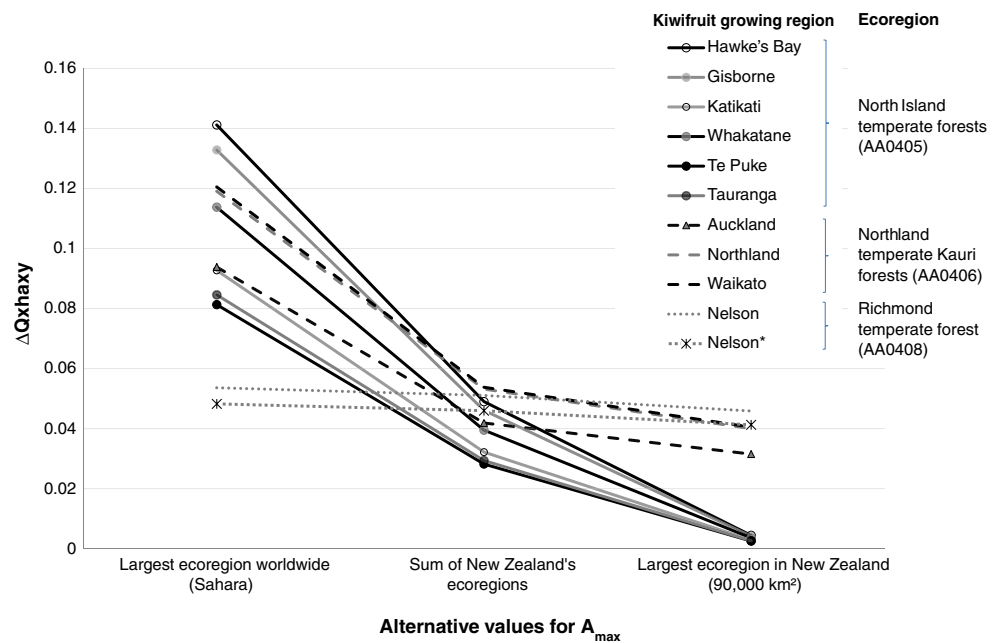
Table 3 Land use impact on biodiversity from kiwifruit plantations using national data for assessing ES and EV on a detailed level. See text for more details

Location	TE/ha	ES	EV	CMB	ΔQ	ha/y	$\Delta Q \times ha \times y$
Nelson	6,509	0.969	0.888	0.3	0.603	0.154	0.093
Te Puke	8,451	0.729	0.661	0.3	0.337	0.118	0.04
Tauranga	8,125	0.799	0.746	0.3	0.417	0.123	0.051
Katikati	7,411	0.823	0.734	0.3	0.423	0.135	0.057
Auckland	7,409	0.831	0.799	0.3	0.465	0.135	0.063
Whakatane	6,040	0.912	0.917	0.3	0.586	0.166	0.097
Northland	5,839	0.849	0.623	0.3	0.370	0.171	0.063
Waikato	5,770	0.715	0.884	0.3	0.442	0.173	0.077
Gisborne	5,173	0.981	0.946	0.3	0.650	0.193	0.126
Hawke's Bay	4,868	0.974	0.974	0.3	0.664	0.205	0.136

TE kiwifruit tray equivalents, *ES* ecosystem scarcity, *EV* ecosystem vulnerability, *CMB* conditions for maintained biodiversity, *Q* biodiversity quality

particular vascular plant diversity (e.g., Muller-Wenk 1998; Köllner 2000; Toffoletto et al. 2007; Schmidt 2008; Goedkoop et al. 2009). Species diversity only captures part of the concept of biodiversity, and the correlation between vascular plant diversity and species diversity for other taxonomic groups is in most cases very low. When impacts on biodiversity from land use and land use changes are to be incorporated in LCA, there is a need for indicators that are ecologically sound and easy to measure. Because the relationship between vascular plant diversity and biodiversity is arguable, the use of vascular plants as a surrogate for biodiversity worldwide seems to only fulfill the need for an indicator that is easy to measure and even then still depends on actual knowledge of vascular species diversity, which is not available for all ecosystems in the world (de Baan et al. 2013).

Our intention has been to demonstrate an alternative approach where impacts on biodiversity can be assessed without the need to use any taxa as indicators. The method this study is based upon was originally developed for boreal forests but is here adapted to be used in any other region of the world, and an example for kiwifruit orchards in New Zealand is presented. Michelsen (2008) introduced an ecosystem-specific index (CMB) and used this to assess the impact on biodiversity in the ecosystem in focus given the identified use of the area. Such CMB indexes must consequently be developed for all ecosystems and scaled for all relevant activities. This requires data that are often not readily available, which is also the case for kiwifruit production in New Zealand. This lack of data is overcome by using the generic values for ecosystem naturalness provided by Brentrup et al. (2002). With this adjustment, the methodology proposed by Michelsen (2008) becomes applicable for almost any land use worldwide, since Brentrup et al. (2002) provide naturalness values for almost all land use situations and Michelsen's (2008) proposal makes use of globally available data on ecosystem scarcity and ecosystem vulnerability at ecoregion level (WWF 2012). The results for the 10 different kiwifruit growing regions of our case study are shown in Table 1. As long as productivity and location of an ecoregion are known, calculations as shown in Table 1 are straightforward.

It is noteworthy that a major argument for introducing the CMB index is that it makes it possible to evaluate different management regimes differently since not just changes in land use, but also modifications and changes in management could be of major importance for biodiversity (Barnes 1996; Haines-Young 2009). For example, Michelsen (2008) showed how even small changes to management regimes could change the results significantly for timber production in a boreal forest. By using generic data, as in this paper, this is not possible since all similar land use activities (such as kiwifruit production) in an ecoregion will result in equal impact, irrespective of management differences, with the only difference being dependent on productivity. We thus face a trade-off between the ability to apply the method globally without a need for new data and the ability to differentiate between different levels of impact between different management regimes.

The sensitivity of central assumptions made by Michelsen (2008) is tested here. To calculate ecosystem scarcity using A_{max}, Michelsen (2008) proposed using the most widely distributed ecoregion (the Sahara) as A_{max}. This follows the same rationale as originally proposed by Weidema and Lindeijer (2001). However, the final score, and consequently the importance of ES for the final value for land use impacts, depends on the choice of A_{max}. When lower values are chosen for A_{max}, ES and ΔQ will also be reduced. This is shown in Table 2 and Fig. 4. This will again influence how the different aspects affect the calculation of total land use impact—as can be seen from Fig. 4, when A_{max} is reduced from the size of the Sahara (4,639,900 km^2) to the area of New Zealand (265,000 km^2) or the most widespread ecoregion in New Zealand (90,000 km^2). When using different values of A_{max}, not only are the absolute scores reduced, but the mutual ranking among the plantation areas is also changed.

When the Sahara Desert is used as A_{max}, EV plays an important role in the kiwifruit case study presented here and clearly separates the Nelson orchard from the others. What really differentiates the other orchards is the land requirement, i.e., the kiwifruit yield (Table 1). With reduced values of A_{max}, land requirements become less important and the ecoregion in which the orchards are located increases in importance.

From our point of view, there is a need to balance the total area requirement to the potential quality of these areas, but

with present knowledge, it is not possible to decide upon what is the correct level. This is, at least partly, a normative question; i.e., what is most important—to reduce the area requirement or to protect specific areas? This simply is a weighting problem: the less weight that is given to quality aspects of the land used, the more weight will be put on area demand, which is directly correlated to yield, irrespective of how quality is assessed.

At this stage, at least for international comparisons, our recommendation is to use ecoregions, using the Sahara as default A_{max}. There are two main arguments for this. First, the problem of lack of data is eliminated since these data are available for a global scale. Second, this also ensures consistency, which at present is hard to accommodate when using and comparing different national datasets. In the kiwifruit case study presented here, the use of Sahara as A_{max} gave small differences in ES for the relevant ecoregions in New Zealand; however, since all ecoregions in New Zealand are relatively small and endemic to the country, and two of them, hosting 9 of the 10 orchards, are regarded to be in a critical state of biodiversity loss, this makes sense from a biodiversity-conserving perspective. However, it is important to have in mind that size of ecoregions is not fixed once and for all; with new knowledge, new subdivisions might occur and also global systems at a finer scale might be available.

Equally important is the choice of structural level. Both Michelsen (2008) and our study (Tables 1 and 2) focused on ecoregions. Again, obvious reasons for this are globally available data and the ability to make international comparisons. The problem is that data, at least for EV, are rather crude at this level and a very large number of areas are termed critical and get the highest score. As a consequence, large areas are treated equally and this is most likely not correct under all circumstances. For example, even though an ecoregion can be regarded as being in a critical condition, there are still structures at lower levels that are not endangered (Lindgaard and Henriksen 2011).

We used the LENZ classification as an alternative to ecoregions to provide a more precise picture of the actual status of the areas in focus. Here, we chose to use LENZ level 4, increasing the number of potential area units in New Zealand from nine ecoregions to 500 LENZ types. For most of the orchards presented here, the use of LENZ indicated a lower impact of kiwifruit production on biodiversity since EV was assessed to be lower in all but one case. For the Te Puke and Northland orchards, the reduction in EV was close to 40 %. Interestingly, EV increased for the Nelson region when a value more specific to that environment was used, further stressing limitations of EV at the ecoregion level.

There were 75 LENZ structures relevant for our case study, and for example, the orchards around Napier occurred in 17 level 4 structures (see the Electronic Supplementary Material). However, when only 3 to 5 of the LENZ environments, covering most of each kiwifruit region, were included, the results did not change much (results not shown). It might thus be possible to obtain similar results using considerable less data; however, we do not attempt to state the minimum number of structures to be included as this would probably be case sensitive. Our assessment was done using publicly available land cover data (LCDB2) from which we determined the amount of orchards in each region, and in which environment from LENZ level 4 they were located. The results are subject to (1) the data quality of the geospatial information used, such as temporal correlation between LCDB2 (which is based on satellite images dated from 2002) and the reference year used in this research and (2) the assumption that the distribution of kiwifruit orchards is proportional to the distribution of orchard and perennial crops (from LCBD2) in that environment. It is thus assumed that the geographic and temporal correlations sufficiently fulfill the aim of the study, providing a general picture of areas that have been mapped as orchards and perennial crops, which are or could be kiwifruit plantations.

Still, we do not recommend this as a general approach. It is important to bear in mind that LCA should be a tool supplementary to other tools. The aim should not be to make LCA so detailed that users could regard it as an alternative to, for example, site-specific assessments for area and species protection or detailed planning (e.g., of precise locations for new plantations). These questions are probably better dealt with using other planning tools. We have shown here that the present EV values at ecoregion level are probably too coarse. The only region not given the highest score in Table 1 (Nelson) has an average value when local data are used (Table 3). The level of detail needed (ecoregion or LENZ level 4) for impact assessment of land use is still an open question.

We also propose a new equation for calculating EV (Eq. (6)). Although this did not have much influence when results were assessed at ecoregion level (Fig. 4), the approach was also used for calculating EV at LENZ level 4 (Eqs. (6) and (7)) and should give a result that is more sound ecologically, but as mentioned in Section 3.2.3, the steepness of this curve can still be debated. A universal curve probably does not exist and a steeper curve would make the differences between EV more pronounced, giving relatively lower values for structures with higher remaining indigenous cover.

A drawback of the use of hemeroby values as a proxy for the CMB index, introduced by Michelsen (2008), is the loss of ability to delineate different management regimes for similar activities in similar areas. It is possible to foresee situations where farmers would want to increase (local) biodiversity within their orchard (e.g., have protected areas set aside). However, this would increase

the area requirement, so the total impact per functional unit might still not change significantly. In this paper, we acknowledge that there is an obvious trade-off between data availability and level of precision of the assessments, keeping the focus on the opportunity to apply the method globally. Another drawback of the hemeroby is the weak scientific basis for the values. The values are assumed to be "an integrative measure of the impacts of all human interventions on ecosystems" (Paracchini and Capitani 2011), but empirical data justifying the scores are mostly lacking. Despite this, they are assumed to be sufficiently robust to be recommended by, for example, the European Commission (cf. Paracchini and Capitani 2011). Also, this is a trade-off. De Baan et al. (2013) have tried to make impacts factors for eight different land use classes on biome level (14 worldwide) and shown data deficits even at this level. Despite their weaknesses, hemeroby values do offer a framework for global assessments applicable to a much finer spatial scale.

Another drawback of the method is the interpretation of the scores in ecological terms. The value is given as quality changes multiplied by area (in hectares) and time (in years). This means that the value 1 represents a total removal of biodiversity in the most valuable structure (ES and EV both 1) over 1 ha for 1 year. The values given are relative to this and changes irrespective of changes in area, time, intrinsic quality and/or impact on biodiversity in the area in focus. As mentioned earlier, the scores will be influenced by the choices and assumptions made; here, exemplified with different scores on ES as a consequence of choice of A_{max} and different scores on EV as a consequence of knowledge of fraction lost of a structure.

6 Conclusions

The goal of this study was to propose a globally applicable method that enables assessments of impacts on biodiversity from land use, using kiwifruit production in New Zealand as a case study. This was done by combining two previously proposed methods, where naturalness data from Brentrup et al. (2002) were used as default values for the CMB index (conditions for maintained biodiversity) proposed by Michelsen (2008) and combined with data on ecosystem scarcity and ecosystem vulnerability. Scarcity and vulnerability are parameters that are intrinsic to the area under study while the CMB index relates to the land use intervention. The CMB index brings to the model the principle of the change of quality caused by the land use. The use of the hemeroby concept as a proxy value for *CMB* is indeed a simplification, but allows the model to be applied globally as long as the area requirement and location in an ecoregion are known (as demonstrated by the kiwifruit case study). If hemeroby's linear approach can limit the use of the model for being generic on the one hand, the concept is easy to apply and does not require any extra data other than yield and location on the other hand.

The sensitivity of the method was assessed by varying A_{max} and by different approaches to calculating EV. This showed that present values for EV at ecoregion level are probably too crude to give a realistic picture of the onsite situation in specific cases. We believe our assessment carried out using LENZ data with a higher spatial resolution gives a more realistic representation of the actual vulnerability of the areas in focus, but agree that such data are not readily available for all locations. Different choices of A_{max} give different weights to the different aspects of land use.

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