Transparent Planning For Biodiversity And Development In The Urban Fringe

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Abstract

In Australia, over 50% of threatened species occur within the urban fringe and accelerating urbanisation is now a key threat. Biodiversity near and within urban areas brings much social benefit but its maintenance involves complex tradeoffs between competing land uses. Urban design typically views biodiversity as a development constraint, not a value to be optimised into the future. We argue that decisions could be more transparent and systematic and we demonstrate that efficient development solutions can be found that avoid areas important for biodiversity. We present a case study in the context of land use change across the City of Wyndham, a local Government west of Melbourne, Australia. We use recent advances in reserve design tools to identify the best tradeoffs between competing values. We suggest that government agencies could adopt similar approaches to identify efficient planning solutions for both biodiversity and development in urban environments.

Introduction

Consistent with a worldwide trend, the size of Australian cities has increased dramatically over the last 100 years (UNFPA, 2007). Increasing numbers of people are choosing to live in urban environments, with approximately 75% of Australians living in the metropolitan areas of capital or smaller cities and this is projected to increase to 90% by the year 2011 (Newton et al., 2001). Rapidly increasing urbanisation rates pose one of the greatest threats to the substantial biodiversity of the urban fringe (Goddard, Dougill & Benton 2010; J. Williams et al., 2001) and create an urgent need to improve conservation planning practices in those areas. The biodiversity of remnant areas proximal to cities is considered nationally and internationally significant, with over 40% of nationally listed threatened ecological communities (Newton et al., 2001) and more than 50% of threatened species occurring in urban fringe areas (Yencken & Wilkinson, 2000). While the literature is clear that the expansion and intensification of human settlement has serious implications for biodiversity (Miller & Hobbs, 2002; Pickett & McDonnell, 1993; Stenhouse, 2004), the loss of natural ecosystems within and adjacent to the limits of a city also poses risks to public health and the quality-of-life of urban citizens (Binning, Cork, Parry, & Shelton, 2001; Boland & Hundhammar, 1999).

Conservation planning in the urban fringe poses many challenges. Firstly, a long-term strategic view is required, as ad-hoc conservation planning efforts will ultimately fail to protect remnant patches of vegetation (Pressey, Humphries, Margules, Vane-Wright, & Williams, 1993) either from outright loss or gradual degradation due to the incremental pressures of urbanisation. Urban development is inherently hostile to nature conservation, as built up areas and their attendant infrastructure are impermeable to the dispersal and movement to a range of organisms.

Secondly, protection of habitat for biodiversity in urban fringe areas involves tradeoffs between a complex range of land uses including housing, industrial development, agricultural production and conservation, and the intensity of the pressures placed on natural areas is often much higher than other regions. The inflated cost of land means that conservation budgets can often be more efficiently allocated elsewhere to achieve conservation objectives. Vegetation cleared for development is often required to be 'offset' by revegetation elsewhere (eg. (Department of Environment and Conservation (NSW), 2005; Victorian Government, 2002). However, the inflated cost of land for revegetation in urban areas tends to direct investment away from peri-urban areas. There are many ecological challenges to implementing offsetting policies including that biodiversity assets are relatively fixed spatially and temporally and, unlike other land uses, cannot be readily transposed from one area to another (S.A. Bekessy et al. 2008).

Despite the introduction of planning legislation and frameworks to preserve biodiversity, many cities around the world are facing a looming extinction crisis; short-term economic gains consistently win over biodiversity concerns on a localised case-by-case basis. The problem of cumulative impacts stems from the difficulty of demonstrating that while each single land use change can have a low overall impact on biodiversity, the accumulation of individual changes over time and within a region might well constitute a major impact (Theobald et al 1997). There is often little scientific input into the biodiversity aspects of the urban planning process and consideration of biodiversity is typically ad-hoc (Bekessy & Gordon, 2007). Frequently, the urban design response to nature conservation is to view biodiversity along with other factors, such as flood risk, as a development constraint, rather than a value to be optimised into the future. Tools such as planning charrettes (Steiner et al., 1999) are often used to incorporate a range of stakeholder views, but the public transparency and democracy of such approaches can be lacking (Margerum, 2005).

Opportunities exist to substantially improve the way that biodiversity is considered in urban planning through the development of tools that optimise the trade-off between conservation objectives and other competing demands of urbanisation within ecological, legislative and policy constraints (A. Gordon et al. 2009). We argue that it is possible to use existing conservation planning tools to transparently and objectively find an efficient urban planning solution that accommodates biodiversity and development. We demonstrate this approach to land use allocation decisions using spatial representations of biodiversity attributes and a spectrum of development scenarios within the City of Wyndham, a municipality on the western fringe of Melbourne. This method builds on recent advancements in ecological modelling and mathematical optimisation to facilitate transparent decisions based on optimal trade-offs between competing values (A Moilanen, 2007; A Moilanen et al., 2005). Maps can be produced that identify areas with high biodiversity and areas of low biodiversity that would be most suitable for development from the perspective of species conservation. Tradeoffs can be then made explicitly by incorporating other social or economic requirements in the optimisation process. The modelling output is spatially explicit and visually compelling, addressing an identified need in urban biodiversity planning (Sandström, Angelstama, & Khakeec, 2006). We do not argue that the tool should be used to determine concrete planning outcomes, but that it should be used to inform the decision-making process in order to achieve more strategic and transparent conservation planning in urban environments.

Methods

The following section outlines the steps taken to create development plans that are spatially optimized for biodiversity while incorporating a range of social and economic requirements. First, we describe the study site, which is a designated growth corridor that contains highly threatened vegetation and species. Second, we describe the development of the various layers that will be optimised, including habitat maps for threatened fauna species, the condition of the vegetation, and layers representing a sample of other elements that planners need to consider, in this case proximity to public transport, flood risk and the cost of maintaining remnant vegetation. Third, we describe the process of finding landscape designs that optimize across these layers using the ZONATION software.

Study Site

The city of Wyndham is located on the south western fringe of the urban extent of greater Melbourne (see map, Figure 1) and has been identified as a key growth area to accommodate future urban expansion (Victorian Department of Sustainability and Environment, 2002). The area is at the eastern extremity of the vast volcanic plain that stretches from the South Australian border region in the west of the state of Victoria to the northern suburbs of Melbourne. The area is characterised by low rainfall and heavy clay soils, which can produce extreme seasonal drought stress particularly in *El Nino* years. This typically results in limited woody tree and shrub growth. Apart from the riparian vegetation associated with the major rivers and streams and a few large freshwater wetlands, the pre-European vegetation of the study area would have been largely treeless.

Lowland temperate grasslands are among the most threatened ecosystems in Australia, with less than 1% of the original extent remaining (Barlow, 1998). The Basalt Plains Grassland Community – to which treeless remnants within the study area belong – is listed as critically endangered under the Commonwealth *EPBC Act 1999*. Threats to the community are current: over 50% of remnants present around Melbourne in 1985 were lost in the following 15 years as a result of continuing urban development and poor management practices (N. S. G. Williams, McDonnell, & Seager, 2005) and losses continue to occur in the rural landscape as a consequence of pasture improvement and cropping. Further, the study area occurs within the Victorian volcanic plains bioregion, which is under-represented by conservation reserves compared to other bioregions around Melbourne (M. McCarthy, Thompson, & Williams, 2006).

Numerous isolated and often highly degraded grassland remnants persist in the heavily developed parts of the eastern section of the study area. Many of these remnants are the legacy of the inability of past planning processes to appropriately accommodate biodiversity requirements. Notwithstanding the ongoing site management issues, some of these reserves retain significant biodiversity and are highly valued by sections of the local community. The study area supports populations of more than 50 state or federally listed fauna species and numerous threatened plant species. Part of the Western Port Phillip Bay (Western Shoreline) and Bellarine Peninsular RAMSAR (Convention on Wetlands) listed site is located within the study area and is therefore considered a site of national significance.

As a designated growth area under *Melbourne 2030* (Victorian Department of Sustainability and Environment, 2002), approximately 30,000 new homes will be constructed in the area over the next 30 years, along with intensive commercial and industrial development.

Habitat Maps

Binary maps were created for each of the rare and threatened fauna species (Victorian Department of Sustainability and Environment, 2003) known to occur within the study area, indicating the presence or absence of 'potential habitat'. Potential habitat was defined as including all land uses and all vegetation and wetland types that may support individuals of the subject taxa. Rules defining each of these binary maps were elicited from various specialist ecologists with local field experience by posing the question, "What land uses and vegetation and or wetland types as defined by the available spatial data, never comprise habitat for this species?" Once this was satisfactorily determined the residual landscape became 'potential habitat'. This data was supplemented with limited field assessments. It is acknowledged that the potential habitat models (syn distribution maps) do not reflect the suitability and viability of habitat for species and populations. In addition, the models have not been subject to any rigorous evaluation and should be considered indicative only, for the purposes of demonstrating the method. See Wintle et al. (2005) for a description of data requirements for more accurate habitat modelling.

Potential habitat or distribution maps were compiled within a GIS, using vector data resolved to 1:25,000 scale, relating to land use, vegetation type, wetlands and watercourses. Maps were built for 32 birds, 4 mammals, 2 amphibians, 3 fish, 4 reptiles, and one invertebrate. The habitats of threatened plant species were not specifically mapped, as the distribution of rare species within the study area is idiosyncratic and closely tied to the specific land use histories and land use intensities that have operated at any particular site. Hence, vegetation extent and condition was used as a coarse surrogate for the distribution of flora species. See Elith and Burgman (2002) for a description of data requirements for more accurate habitat modelling of rare plants.

Mapping Vegetation Extent And 'Habitat Condition'

As much of the study area is privately owned land, existing vegetation mapping, which has historically focussed on public land, proved to be inadequate. Therefore, remnant vegetation across Wyndham was mapped employing field reconnaissance, Aerial Photograph Interpretation (API) of recent 1:5,000 scale digital aerial photography and advice from Government agency officers, environmental consultants and local naturalists. Vegetation was classified in accordance with the established Victorian typological framework (Victorian Government, 2002). Line work was digitally captured and subsequently ground-truthed.

A surface representing 'habitat condition' was generated, ranking the entire study area on the basis of observed site attributes measured against an appropriate archetype or benchmark and landscape attributes. A full description of this benchmarking approach, including the attributes employed and their weightings within a combined condition index, is provided in Parkes et al. (2003). Scores for structure and composition (score range 1-25) and the relative abundance and dominance of exotic weeds (score range 1-15) were allocated to homogeneous 'patches' of vegetation. Constraints to site access precluded a detailed appraisal of the condition of vegetation and habitat, and sites supporting native vegetation were largely assessed from roadsides. Simple landscape attributes were generated for patches of vegetation within a GIS, including patch size (range 1-15), scaled density of habitat/vegetation (range 1-10) and distance to the core area of a large local remnant (score range 1-5) (Parkes et al., 2003).

Cost Layers

Figure 2 presents a set of cost layers that were developed for use in the various planning scenarios. Biodiversity cost (Figure 2(a)) was calculated using the 'landscape context' GIS layer (DSE Corporate library; Wilson and Lowe (2003)), which represents the condition of the site and connectedness to other vegetation within the region. We assumed that sites with a lower landscape context score would be more costly to restore and maintain, hence these sites were allocated a lower value for biodiversity (and a higher potential value for development). Figure 2(b) presents biodiversity cost added to flood cost, which assumes a lower potential value for development in flood-prone sites. Figure 2(c) includes proximity to rail, whereby potential value for development decreases with distance from the existing rail line (proximity to the railway line was used, rather than proximity to railway stations because new stations are proposed under the planning document *Melbourne 2030*). Areas were weighted by their perpendicular distance from the rail line according to Table 1. Figure 2 (d) presents a cost layer that is a sum of the three cost layers, biodiversity, flood risk and rail line proximity.

Spatial Optimisation

The objective of the optimisation process is to select an arrangement of habitat patches that maximizes habitat quality, species richness and rarity, while maintaining habitat connectivity. Economic and social factors were also included as potential 'costs' or 'benefits' to be optimised. The optimisation procedure was conducted for a range of scenarios given different proportions of the landscape available for habitat protection (see Table 2).

Landscape solutions were calculated using the ZONATION method and software (A Moilanen, 2007; A Moilanen et al., 2005; A. Moilanen & Kujala, 2006). ZONATION ranks all cells in the landscape according to representation of biodiversity features, complementarity and the degree of habitat connectivity. The algorithm considers the full landscape to start with, and then iteratively discards grid cells of lowest value from the edge of the remaining area, thus maintaining a high degree of structural connectivity in the remaining habitat. (The condition of removal from edge improves computational efficiency, but it can be relaxed if so wished). The Zonation algorithm differs from target-based planning or maximum coverage reserve selection (see Moilanen (2007) for details). Instead of finding a single optimal solution, such as the least expensive set of sites that achieves targets, it generates a hierarchy of solutions. The hierarchy is generated via a strategy of minimization of marginal loss, the iterated removal of that cell whose loss causes the smallest decrease in the conservation value of the remaining reserve network. Thus, instead of a single selection of sites, it generates a gradation of conservation priority throughout the landscape (such as Figure 3(a)) and an associated set of curves (such as Figure 9), describing how well each species (or land cover type) does at any given level of cell removal. Specification of species weights, connectivity requirements and the so-called cell-removal rule result in a balanced species representation at each level of landscape availability. The hierarchical structure of the solution means that the best 1% is within the best 2% of the landscape which is within the best 5%, and so on, which allows for easy visualisation of results. Any given top fraction of landscape can be simply identified after a ZONATION run, because the removal hierarchy of cells is saved. Likewise, any given least useful fraction of the landscape can be identified, which was the objective in this study – to identify areas most suitable for urban development. The nature of the ZONATION algorithm allows it to be run on data sets in the order of millions of landscape elements (grid cells) combined with hundreds of species, which facilitates a direct link between statistical habitat suitability modelling on GIS grids and ZONATION.

The ZONATION meta-algorithm, as given by Moilanen (2007) is simple: (1) Start from the full landscape. Set rank $r = 1$. (2) Calculate marginal loss following from the removal of each remaining site i, δ_i. (3) Remove the cell with smallest δ_i, set removal rank of i to be r, set r=r+1, and return to 2 if there are any cells remaining in the landscape. The critical part of the algorithm is the definition of marginal loss, where many complications can be introduced. These include techniques for generating reserves that have been aggregated in a species-specific manner, distribution smoothing (A Moilanen et al., 2005; A Moilanen & Wintle, 2006) and the boundary quality penalty (A Moilanen & Wintle, 2007). The algorithm allows uncertainty analysis, aiming at robust reserves that are likely to contain the species (A Moilanen et al., 2006; A Moilanen & Wintle, 2006). The technique of replacement cost analysis (Cabeza & Moilanen, 2006) can be used to evaluate the conservation value of an unconstrained optimal solution against solutions that either forcibly include proposed/existing reserve areas or forcibly exclude areas required for agricultural-urban development. The ZONATION software and a user manual (A. Moilanen & Kujala, 2006) are freely available via the website (www.helsinki.fi/science/metapop).

There are three basic alternatives for the so-called cell removal rule, used in step (2) of the ZONATION meta-algorithm, namely core-area, additive benefit function and targeting benefit function. Each of these corresponds to slightly different assumptions about the planning objective, how local quality is valued and how biodiversity features are traded off against eachother. In this study we used the core-area algorithm (A Moilanen, 2007; A Moilanen et al., 2005). It has the properties that (i) species weights and land cost are included in prioritisation,

(ii) high-quality locations are preferred for all species even if the occurrences are in species-poor areas. Compared to additive benefit functions or target-based planning, the core-area algorithm generally produces solutions which have lower average representation levels, but higher minimum representation across species and higher local quality for selected locations (A Moilanen, 2007).

Technically, the core-area algorithm defines marginal loss caused by the loss of cell i as:

$$\delta_i = \max_j \frac{Q_{ij}(S)w_j}{c_i}, \qquad \text{(equation 1)}$$

where w_j is the weight of species j and c_i is the cost of adding cell i to the reserve network. The weight can be used to prioritise species according to, for example, their taxonomic uniqueness or some measure of global rarity. Cell cost can be any measure of (opportunity) cost following the allocation of the cell for conservation – here cost was related to flood proneness or proximity to railway.

The critical part of the equation is $Q_{ij}(S)$, the proportion of the remaining distribution of species j located in cell i in the remaining set of cell, S. When a part of the distribution of a species is removed, the proportion located in each remaining cell goes up. This means ZONATION tries to retain core areas of all species until the end of cell removal even if the species is initially widespread and common. The min-max facilitates the algorithm feature that occurrences are not treated as additive, but that high-quality locations are strongly preferred for species. Figure 3 illustrates the workflow we used with ZONATION in this study.

Results

A map of the growth area prioritised for the biodiversity attributes only (scenario 1) is presented in Figure 4. Cells are ranked 0-1, where a value of 0.98 would indicate that 98% of cells would be removed from the landscape before that cell would be chosen for development. Several ‘hotspots’ can be identified, as well as areas more preferable for development.

Optimised solutions for a range of development scenarios are presented in Figures 5-8. The red areas represent those that would be chosen for development and the remaining cells are graded light to dark, with darker areas indicating higher priority for biodiversity. In each figure, (a) represents the ZONATION output, ranking cells from highest value to lowest value; (b) represents the difference between rankings for the scenario compared with figure 5 (biodiversity only), where lighter areas represents cells that have increased in their ranking, and darker areas have decreased in their ranking; and (c) represents the lowest ranked 10% of the landscape, which could be deemed most suitable for development (apart from Figures 4 and 8 which only displays ZONATION output and lowest ranked 10% of the landscape).

Figure 9 presents plots for each scenario of the proportion of the landscape lost against the minimum proportion of habitat available to any of the species modelled. This figure describes how robust any given level of cell removal is to the species that suffer the greatest (proportional) loss of habitat. Overall, it is apparent that 10% of the landscape could be developed with relatively minor (4% average) biodiversity loss, and that the differences between scenarios are small in this respect. Thus, flood-prone areas could be avoided and proximity to rail preferred with minor biodiversity consequences. The only significant difference between scenarios is between the market-gardens scenario (scenario 5; Figure 9d) and other scenarios. In scenario 5 more land is available for development, and the development of the market-gardens has little impact on biodiversity, which is apparent from Figure (9d) as the loss of 10% of the landscape results in close to zero biodiversity loss. In all scenarios the influence of flood avoidance or proximity to rail is only apparent at high levels of habitat loss (not shown).

Discussion And Conclusions

This case study demonstrates a method that can improve rigour and transparency in urban planning, while incorporating scientifically derived criteria for biodiversity conservation. The process involves gathering data, identifying and weighting key values according to stakeholder preference, and modelling to produce visual representations of possible scenarios that have been optimised according to the chosen values.

The method confers several advantages to the planning process. Firstly, it recognises that the ecological foundations of a site are less portable than other considerations (Fallding, 2004). The modelling method provides a mechanism for making tradeoffs in the least harmful way for biodiversity, incorporating the spatial distribution of biodiversity early on in development planning. Setting biodiversity as an underlying value to be optimised encourages tradeoffs to be made in a more timely and transparent way.

Secondly, it encourages decision-makers to explicitly rank priorities. The objective function for the optimisation can be decided upon using a democratic process, whereby stakeholders openly debate and decide upon appropriate weightings for competing values. The implications of different weightings for biodiversity conservation, or different valuation philosophies (van der Windt, Swart, & Keulartz, 2007) can then be explored. In addition, the tool provides opportunities for the community to be exposed to the complexities and consequences of land use.

This could serve to further democratise both the planning process and the planning outcomes and increase the level of public transparency. The tool provides powerful visual representations of the planning scenarios that can be used to integrate objectives and explore tradeoffs.

Thirdly, the tool highlights dilemmas between competing objectives and encourages discussion of the implications of different tradeoffs. In the Wyndham case study, several competing sustainability objectives were explored. The dilemma of prioritising biodiversity conservation over public transport-oriented development was examined. The spatial configuration of least valuable cells was identified while varying the weighting given to the competing objectives. A further sustainability dilemma was highlighted between biodiversity protection and local food production. The development scenarios were initially developed masking out the area currently used for market gardens, as these areas were deemed commercially valuable and hence unavailable for housing development. Furthermore, local food production has emerged as a significant priority for greenhouse gas reduction. However, if biodiversity was the major community concern in the region, and food production was a low priority, the market gardens would be designated for development as the areas of least impact on biodiversity (Figure 9).

The implications of alternative biodiversity conservation policies can also be explored. For example, a 'triage' approach would weight critically endangered species higher than less threatened species (M. C. Bottrill et al. 2008). Alternatively, least endangered species would be prioritised if the focus biodiversity policies aim to preserve those species that are most likely to become critically endangered in years to come (McIntyre, Barrett, Kitching, & Recher, 1992). Policy choices of this kind are typically made without explicitly exploring the implications of different trade-offs (McIntyre et al., 1992).

Finally, the exercise can also lead to the identification of alternative development approaches that could reduce environmental impact. For example, despite some changes in sub-division size, Melbourne's peri-urban regions continue to represent housing densities significantly below those employed in cities of international comparison (Scheurer & Buxton, 2005). If the average housing density could be increased even slightly, the area required to fit 30,000 new homes would be substantially reduced (Scheurer & Buxton, 2005). While this tool provides a transparent mechanism for articulating tradeoffs in urban planning, it does not indicate whether decisions are ultimately 'acceptable'. The decision to clear habitat to meet competing objectives is a social one, but should be made acknowledging the risks to environmental and other concerns. A decision theory framework that articulates costs, benefits and risks could be useful in this context (Possingham, 2001).

A key limitation of the case study presented here is that the quality of the landscape optimisation depends on the quality of the underlying data. In this scenario, the species distribution models and habitat quality assessments undertaken may not be adequate or sufficiently accurate surrogates of the region's biodiversity to appropriately inform the allocation of land use (see M. A. McCarthy et al., 2004). Data quality was further reduced by access constraints restricting surveys on private property. Error and uncertainty in underlying GIS maps has been shown to translate into 'inefficient, unrealistic or erroneous' land management decisions (Rae, Rothley, & Dragicevic, 2007). Ideally landscape optimisation within the urban context would be informed by spatially and statistically explicit models of the habitat and potential for persistence of the entire indigenous biota. Such a data set would require a substantial and possibly unrealistic amount of additional genetic and biophysical inventory, modelling and research. In addition, its assembly would require an extended lead up period before the decision making process to ensure that seasonal detectability issues typical of grasslands are addressed (Garrard, Bekessy, & Wintle, 2008). In urban areas, biodiversity is routinely considered at the project assessment phase when decisions about spatial arrangement have already been made (Fallding, 2004). Identifying the minimum biodiversity data set required to make robust decisions for biodiversity conservation and analysing of the impact of underlying uncertainty (A Moilanen et al., 2006) on the selection of priority areas for conservation in the peri-urban context will be the focus of further research.

The modelling tool presented here assists in identifying areas that best represent a sub-set of the species present in the region, but the tool tells us little about the likely persistence of those species into the future. Maintaining the viability of species requires consideration of a multitude of factors including landscape elements (such as the fragmentation of habitat and the size and shape of remnants and the types of land uses being carried out with in the matrix), and the requirements of individual species (such as mode of dispersal, rate of replacement and response to urban impacts). Although a simple concept, incorporating ‘species viability’ in the evaluation of conservation planning options remains a significant challenge to conservation planners (Margules & Pressey, 2000). Nevertheless, the method proposed here is a step towards adapting conservation planning methods to planning of urban development zones. The approach is novel in that we use ‘reserve design’ tools in an inverse manner to identify areas of least impact on biodiversity assets that are consequently preferable for development.

Acknowledgements

This research was conducted with the support of funding from the Australian Government’s National Environmental Research Program, the Australian Research Council Centre of Excellence for Environmental Decisions and the Australian Research Council under linkage grant LP0882780 with the following industry partners: Department of Sustainability and Environment, Victoria, Parks Victoria, Hume City Council, Mornington Peninsular Shire andThe Port Phillip and Western Port Catchment Management Authority. Thanks to Natalie Davis, Rachel Bekessy and two anonymous reviewers for comments on the manuscript. Sarah Bekessy conducted this research as a Brian Robinson Fellow, which funded research leave in Seville at the Insituto Donana, Sevilla (Dr Jordi Bascompte). A.M. was supported by funding from the Academy of Finland.

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List Of Tables

Table 1. Weightings used to create the cost layer representing proximity to the existing rail line.

Distance from the rail line	Weighting
<2 km	4
2-3 km	3
3-5 km	2
>5 km	1

Table 2 Description of the development scenarios used in the optimisation procedure, including the variables maximised and the weighting used to prioritise values.

Scenario	Objective of prioritisation	Weighting
Scenario 1: Priority given to biodiversity only	Maximal balanced species representation according to habitat maps for 50 listed fauna species. Maximize habitat quality (ranked using the habitat hectares approach (Parkes et al., 2003).	Critically endangered x 10 Endangered x 5 Vulnerable x 3 Rare x 1 Habitat quality x 10
Scenario 2: Biodiversity maximised, while priority also given to allowing development near existing railway	Efficient biodiversity representation away from railways as implemented via the cost layer which was scaled by proximity to rail. Biodiversity as per scenario 1.	Biodiversity layers weighted as per scenario 1. Equal weighting given to proximity to railway
Scenario 3: Biodiversity maximised, while priority also given to development near existing railway, and away from flood prone areas	Efficient biodiversity representation away from railways and flood-prone areas. Biodiversity as per scenario 1. Proximity to railway as per scenario 2. Areas with highest risk of 1 in 100 year flood ranked lowest.	Biodiversity layers weighted as per scenario 1 Proximity to railway and flood risk weighted equally
Scenario 4: Same as scenario 3	Same as scenario 3	Same as scenario 3, with five times greater weighting given to proximity to railway
Scenario 5: Areas current used as market gardens made available for development	Same as scenario 1	Same as scenario 1

List Of Figures

Figure 1 Map of the study area (reproduced with permission from the Victorian Department of Sustainability and Environment, source www.dse.vic.gov.au/planningschemes/wyndham/home.html).

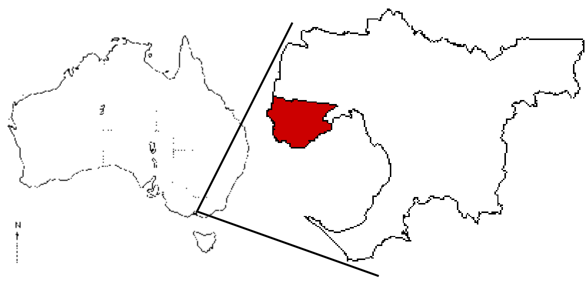

Figure 2. Layers created to represent a range of potential costs to be included in the optimisation under various scenarios.

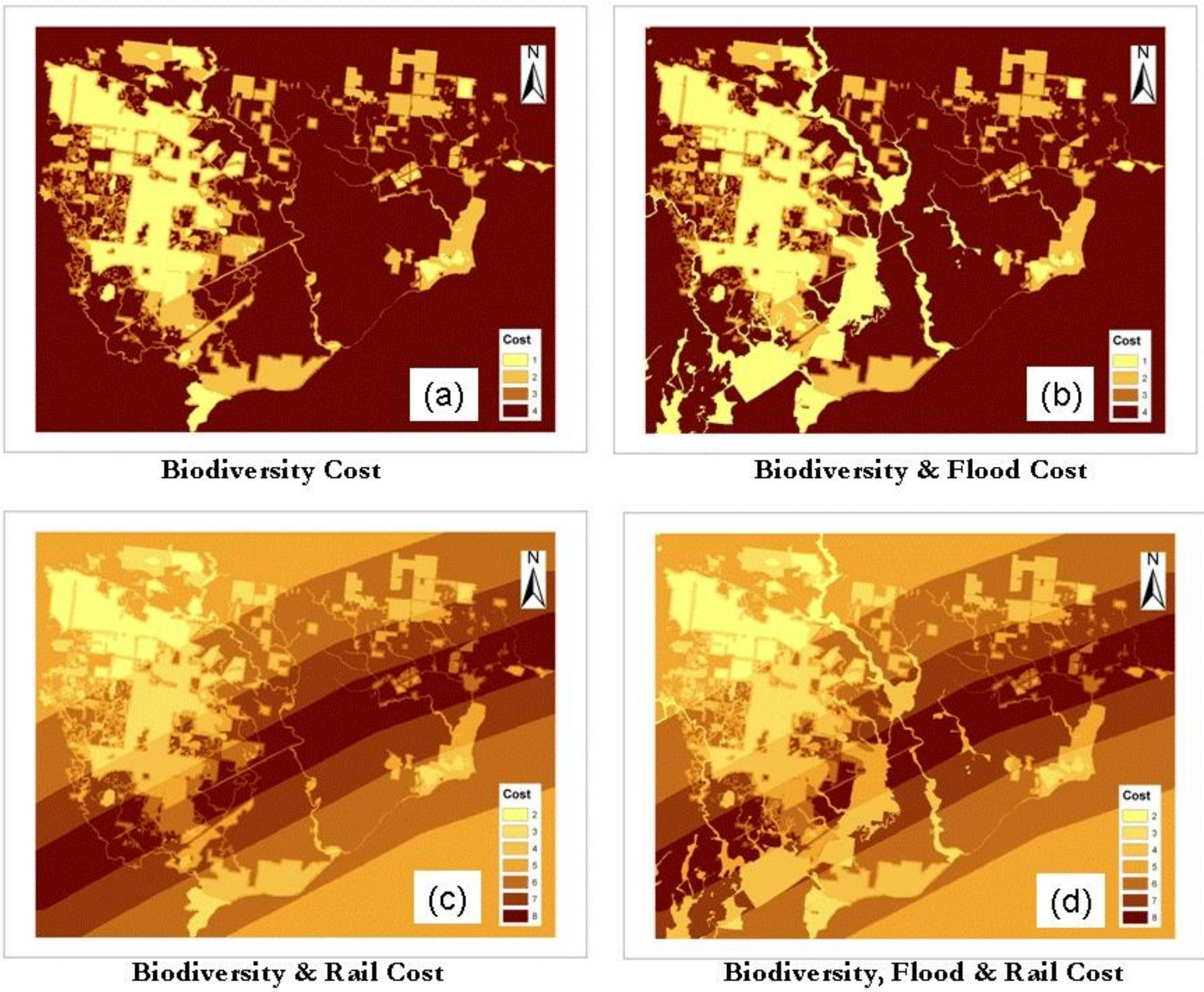

Figure 3 The analysis workflow used in this study. Boxes with thick lines and thin lines represent inputs and analyses, respectively.

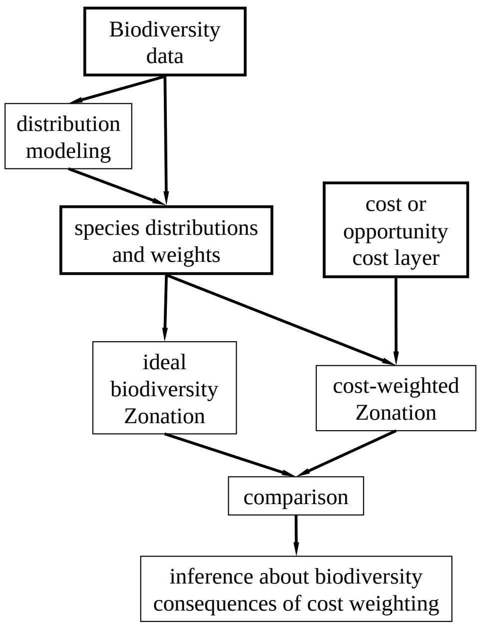

Figure 4 (a) Map of the study area optimised under scenario 1; **(b)** Areas highlighted in red represent the lowest ranked 10% of the landscape, and would be most preferred for development if biodiversity was the only development consideration.

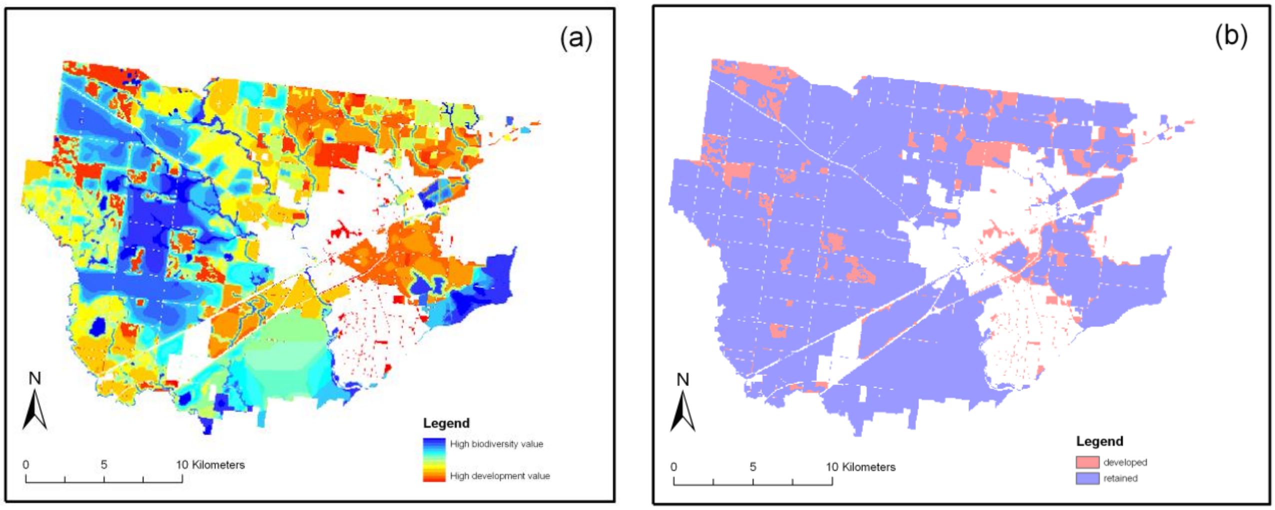

Figure 5 (a) Map of the study area optimised under scenario 2, which weights areas in closest proximity to the railway as being more desirable for development **(b);** The difference plot **(c)** indicates that areas closer to the rail line have decreased in their biodiversity priority ranking, while those further away have increased values.

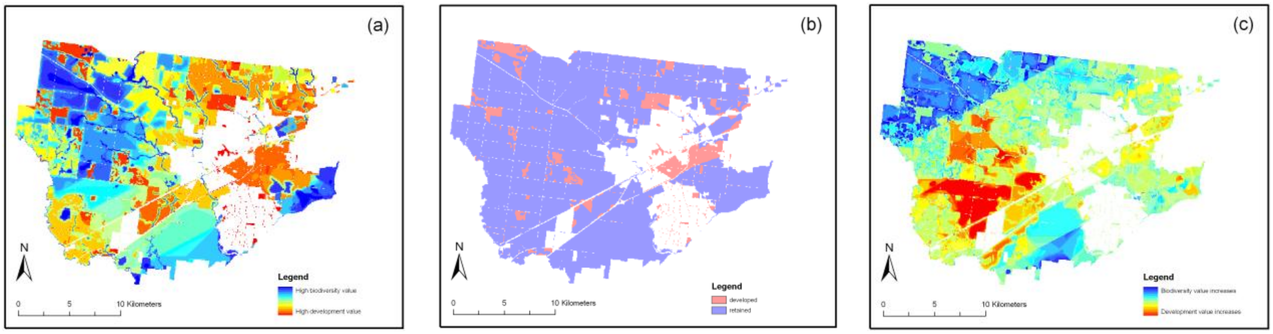

Figure 6 (a) Map of the study area optimised under scenario 3, which considers biodiversity, proximity to rail and flood risk; **(b)** Areas highlighted in red represent the lowest ranked 10% of the landscape; **(c)** difference between scenario 1 and 3 showing that flood prone areas and cells around waterways have increased in their ranking for biodiversity.

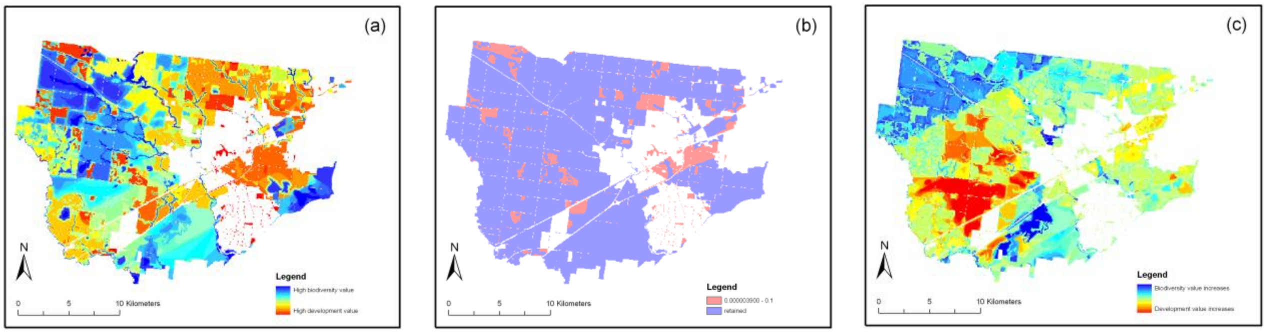

Figure 7 (a) Map of the study area optimised under scenario 4, which weights proximity to rail as a much higher priority; **(b)** Areas highlighted in red represent the lowest ranked 10% of the landscape **(c)** difference between scenario 1 and 4, showing that some other riparian areas have been removed.

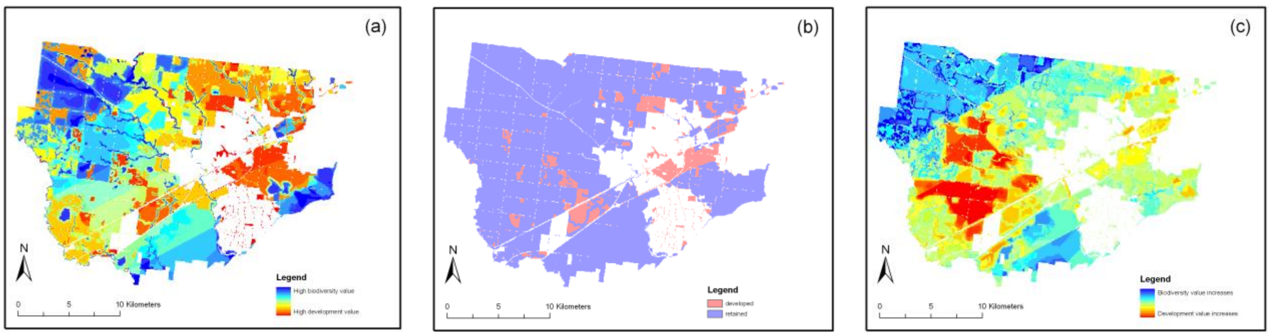

Figure 8 (a) Map of the study area optimised under scenario 5, which allows the market gardens to be considered for development; **(b)** Areas highlighted in red represent the lowest ranked 10% of the landscape

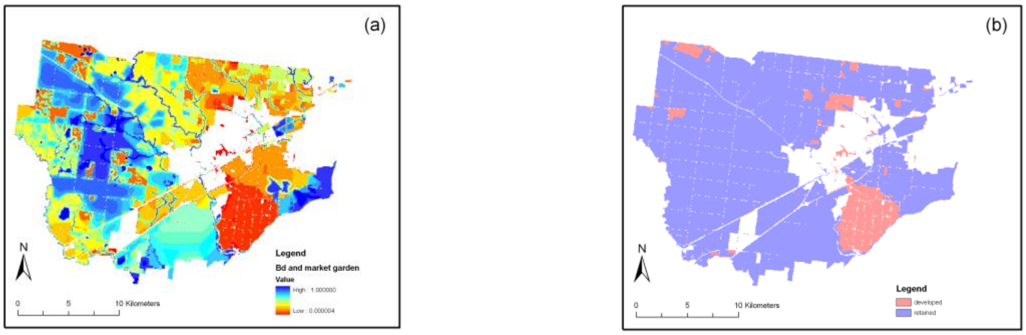

Figure 9 Plots of proportion of the landscape lost against the minimum proportion of habitat available to any of the species modelled. These figures describe how robust any given level of cell removal is to the species that suffer the greatest (proportional) loss of habitat.