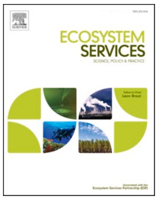
Ecosystem Services 61 (2023) 101527

Contents lists available at ScienceDirect

Ecosystem Services

journal homepage: www.elsevier.com/locate/ecoser



Full Length Article

A multiscale perspective on how much wetland restoration is needed to achieve targets for ecosystem services

Stephanie Tomscha [a,*], Bethanna Jackson [b], Rubianca Benavidez [b], Mairéad de Róiste [c], Stephen Hartley [a], Julie Deslippe [a]

[a] School of Biological Sciences, Victoria University of Wellington, New Zealand
[b] School of Geography Environment and Earth Sciences, Victoria University of Wellington, BEEA Ltd, New Zealand
[c] School of Geography Environment and Earth Sciences, Victoria University of Wellington, New Zealand

ARTICLE INFO

Keywords:
Nitrogen
Phosphorus
Agriculture
Catchment modelling
Percentage targets
Spatial scale

ABSTRACT

Percentage-based targets for conservation and restoration provide a compelling narrative for enhancing nature and human wellbeing. However, evidence is still lacking for what these percentage targets should be, especially for improving multiple ecosystem services. Furthermore, restoration targets can be challenging to implement across decision-making scales. We explored these challenges in the Ruamahanga Basin in the North Island of Aotearoa New Zealand, where ~98% of wetlands have been drained. We created restoration scenarios that increase the percentage area of wetland restored at two spatial scales: first, across the entire Ruamahanga Basin, and second, using subcatchments delineated from the contributing areas of individual historical wetlands. These scales allow us to adopt the decision-making perspective of basin-scale managers/planners as well as groups/individuals restoring single wetlands. At each scale, we estimate gains and losses towards plausible targets for nitrogen and phosphorus retention, carbon storage, and agricultural productivity using the high-resolution, ecosystem service modelling tool, LUCI. At the basin scale, ecosystem service changes were incremental, showing linear trends through to full wetland restoration. We found no percentage value at which restoration did not add benefits for nutrient retention and carbon storage and no percentage value at which restoration did not detract from agricultural productivity. Within subcatchments, gains in phosphorus retention were achieved across all restoration percentages. Nitrogen retention targets were mostly met when the percentage of wetlands restored exceeded 60%. Contrasting outcomes at two different scales showed that most of the variability in ecosystem service outcomes is found at fine spatial scales, rather than at the basin scale, which has implications for choice of policy mechanisms and spatially-targeted management of ecosystem services.

1. Introduction

Percent-area targets, which specify conservation or restoration targets based on portions of land area, are widespread in conservation and resource management. The IUCN's and Global Biodiversity Framework (Target 3) recommendation of protecting 30% of global land and seascapes by 2030 is fundamental to current strategies for conservation (IPCC, 2022; Woodley et al., 2019, https://www.cbd.int/article/cop15-final-text-kunming-montreal-gbf-221222). Percent-area targets offer appealing simplicity for policy, but important questions remain about the effectiveness of these targets (Carroll and Noss, 2021; CBD, 2020; IPBES, 2019). Currently, only 16% percent of global land is protected (Carroll and Noss, 2021; UNEP-WCMC, 2021), but this percentage varies by country. Globally, only 10% of the current protected area network is structurally connected (Ward et al., 2020), and shortfalls in both representativeness of different ecosystems and structural connectivity suggests restoration is needed to fill these gaps.

Setting percent-area targets to improve multiple ecosystem services is challenging, and recommended percent-area targets remain contested in the ecological literature (Banks-Leite et al., 2021). Supporting these recommendations is a long history of research on how much restoration is needed to improve individual ecosystem services, such as phosphorus retention (Wang and Mitsch, 1998). For example, landscape simulations show the percent of agricultural land cover influences nutrient loading

---

* Corresponding author.

E-mail addresses: stephanie.tomscha@gmail.com (S. Tomscha), bethanna.jackson@vuw.ac.nz (B. Jackson), rubianca.benavidez@vuw.ac.nz (R. Benavidez), Mairead.deRoiste@vuw.ac.nz (M. de Róiste), Stephen.Hartley@vuw.ac.nz (S. Hartley), Julie.Deslippe@vuw.ac.nz (J. Deslippe).

https://doi.org/10.1016/j.ecoser.2023.101527
Received 3 May 2022; Received in revised form 30 March 2023; Accepted 3 April 2023
Available online 20 April 2023

in streams, but above 30% agriculture and <50% forest, these outcomes are highly sensitive to spatial patterns of the land use (Gergel, 2005; Thomas et al., 2020). Targets for productive landscapes have suggested that an area of at least 20% native habitat can improve nature's contributions to people and connectivity (Garibaldi et al., 2021). Keeping wetland extent between 3 and 7% of total watershed area is recommended for water quality and flood abatement, but specificity is lacking for these targets in different ecosystem types, such as diverse wetland types (fens, swamps, etc.) (Mitsch and Gossilink, 2000). Moreover, when multiple values and objectives are considered, protected area targets often become high (>50%) (Woodley et al., 2019).

Debates on percent-area restored targets have acknowledged that ideal targets may vary with spatial scale and context. For example, recent claims that forest area should be at least 40% are seen as lacking evidence, and regionally defined percent-area targets are proposed instead (Arroyo-Rodríguez et al., 2020; Banks-Leite et al., 2021). Fractal approaches, whereby a fixed percentage at each spatial scale is restored, have also been proposed (Ekroos et al., 2016). Challenges for setting targets include that spatial scale can influence the detection of ecological thresholds (Spake et al., 2022). While outcomes from restoring different percentages of the landscape vary with spatial scale, quantitative comparisons of ecosystem services after implementing percent-based targets have rarely been examined at multiple scales. Furthermore, restoration decisions in the real world are made at different scales: from broad-scale national policies, through planning by regional councils down to local on-farm management for restoring individual wetlands. Despite spatial scale underpinning how decisions are made in restoration, there is very little work exploring the dynamics of different spatial scales in restoration planning (Gilby et al., 2021). Approaches that explore the consequences of local *vs.* broadscale planning on ecosystem services are sorely needed.

Representation of wetlands, and other at-risk ecosystems, in percent-area targets is important to protect the biodiversity and ecosystem services they provide (Zedler and Kercher, 2005). Globally, between 64 and 71% of wetlands have been drained since the1900′s while in some countries, such as Aotearoa New Zealand, this figure exceeds 90% (Ausseil et al., 2008; Davidson, 2014). In Aotearoa, this wetland loss is especially concerning as wetlands are held in the highest regard by indigenous Māori, being important sources of food, fibres, and cultural services (Taura et al., 2017). Competing with agricultural production in fertile lowlands, wetland restoration can enhance a wide range of ecosystem services, increasing biodiversity and carbon storage, improving water quality and aesthetics, and reducing floods (Clarkson et al., 2013; Tomscha et al., 2021). Furthermore, despite that in Aotearoa, 33.4% of land is already managed by the Department of Conservation (Ministry for the Environment, 2010), loss of native species remains a problem (Hare et al., 2019). This continuing species loss is likely due, in part, to the lack of representation of wetlands and lowland ecosystems in the conservation estate. Protection of existing wetlands provides insufficient area to sustain their functions and services, and specific information on appropriate percent-area targets for wetland restoration are needed.

Much of the research on percent-area targets for wetland restoration have focused on optimizing the placement of wetlands (Newbold, 2005). While optimal wetland placement can improve water quality (e.g., Singh et al., 2019), to date, most wetland restoration in Aotearoa New Zealand occurs opportunistically in locations selected by willing landowners or proximal community groups. Sweeping guidelines, such a 30% protection by 2030 (CBD, 2020; UNEP, 2020), assume broad scale planning and prioritization. There are potentially enormous benefits to broad-scale restoration planning, with estimates of tripled gains and halved costs when targeted approaches are employed (Strassburg et al., 2019). While strategic restoration may produce optimal results (Newbold, 2005; Singh et al., 2019) opportunistic projects may also provide critical momentum for restoration, especially in contexts where there are limited economic incentives for landowners to restore. For example, in Aotearoa New Zealand, voluntary opportunism, alongside stock exclusion regulations (RMA, 2020 i.e., fences required around natural wetland and streams >than 1 m in width) are the primary drivers of the extent and location of wetland restoration initiatives (pers. obs). Given the current lack of coordination, new insights to the simultaneous benefits of restoration, from both the individual decision-making perspective, as well as from a broader basin-scale planning perspective, are urgently needed (Ruamahanga Whaitua Committee, 2018).

Several policy objectives point towards restoring wetlands in Aotearoa New Zealand. As the UN decade of restoration begins, Aotearoa has the responsibility to restore wetlands as a signatory of the Ramsar Convention on Wetlands (Myers et al., 2013). Concomitantly, investment in restoration is growing rapidly through the One Billion Trees programme and on private farms through legislated changes to freshwater protection (Forestry, 2018; Parliamentary Counsel Office, 2020). Thresholds for nitrate levels in drinking water are important as high nitrate levels can contribute to colorectal cancers (Richards et al., 2021), hence, wetlands may improve health outcomes through water denitrification. Alongside water quality targets, Aotearoa New Zealand has obligations to meet targets for mitigating greenhouse gas emissions (Paris Agreement to the United Nations Framework Convention on Climate Change, 2015). Wetlands are known carbon storage hotspots, because vegetation often grows vigorously in wetlands due to high nutrient levels, and anoxic wetland soils reduce decomposition of organic matter, favouring carbon accrual (Bentley et al., 2022; Nahlik and Fennessy, 2016). In summary, wetland restoration provides opportunities for meeting the multiple objectives of regulating the climate, ensuring clean rivers, mitigating flood risk and supporting human health.

While percent-area targets are often explored in simulated landscapes, understanding these dynamics in real catchments underscores how different contexts can change restoration outcomes. In this study, we examined the relationship between the percentage of wetland restored at two spatial scales and ecosystem service change. Our work is focussed on a watershed that has lost all but 2% of its original extent of wetlands, and where regional governments have suggested targets of increasing this extent to 10% of their original extent (Crisp, 2020). We use the LUCI framework and freely available models, to explore changes in four ecosystem services (nitrogen (N) and phosphorus (P) retention, carbon storage, and agricultural production). We ask several questions to better understand the outcomes for restoring different amounts of wetland. 1) How much wetland restoration is needed to achieve water quality standards for N and P? 2) What co-occurring gains will this percentage of restoration achieve for carbon? 3) How much productive agricultural land is given over to restoration in each scenario? 4) Do gains and losses of ecosystem services accelerate, decelerate, or change proportionately with percentage area restored? Our approach also allows us to explore how percent-area restoration targets may lead to different ecosystem service gains and trade-offs, when applied at two different decision-making scales: 1) individual wetland-derived sub-catchments and 2) full basin extents; enabling a more integrated understanding of the benefits of wetland restoration both locally and regionally.

2. Methods

2.1. Study site

In the southern part of Aotearoa New Zealand's North Island, the Ruamahanga Basin (Fig. 1) is traditional territory of two Māori iwi (tribes): Ngāti Kahungunu ki Wairarapa and Ngāti Rangitāne o Wairarapa. Māori farms make up 3.6 percent of all farmlands in New Zealand (Stats NZ, 2021). Contemporary land use is primarily agriculture, including sheep and beef cow farming, dairy and viticulture. Historically, wetlands comprised 26% of the basin, however, wetland loss in this 3360 km^2 basin has been severe with more than 98% (~856 km^2) of

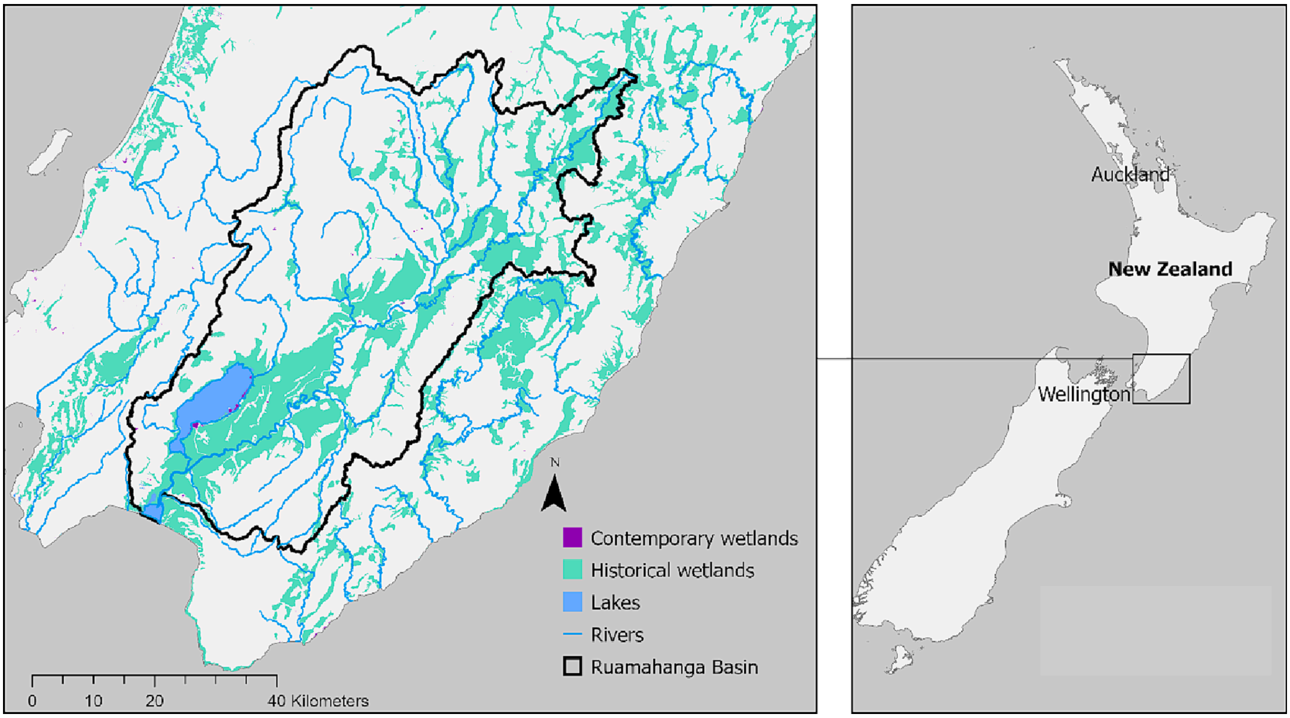


Fig. 1. The Ruamahanga Basin situated on Aotearoa New Zealand's North Island is a 3360 km^2 basin historically rich with wetlands. The thick black line outlines the basin. Approximately 26% of the basin was wetland prior to colonial drainage for agriculture. Green areas show historical wetlands (pre-human), while purple areas show the sparse remaining wetlands.

the historical (pre-human) wetland extent drained (Tomscha et al., 2019). Wetland restoration projects in the basin are starting to reverse this trend; driven by iwi, grass-roots community efforts, farmers, and local government. More than 50 community groups have united to support restoration in the region (https://waip2k.org.nz/, Accessed 22 Mar. 2022). The iwi are strong proponents of restoration, as wetlands were a key food source and formed the backbone of their economy prior to colonization and widespread wetland drainage.

2.2. *Development of wetland restoration scenarios*

The extent and location of historical wetlands guided our modelling of restoration scenarios. The boundaries of the historical wetlands were estimated using a combination of soil data and digital elevation models (Ausseil et al., 2008), available online (https://data.mfe.govt.nz/table/52541-estimated-contemporary-and-pre-human-wetland-area-by-type-2008-estimate/). A wetland was considered restored in our model by switching the land cover from its contemporary cover to indigenous forest. Because historical wetlands comprised $\sim$26% of the Ruamahanga Basin, we are unable to model how wetland restoration benefits water quality at greater percentages, as we assume this to be the maximum possible extent of wetlands in the Ruamahanga Basin. By contrast, via our subcatchment scale approach, we were able to model the benefits of wetland restoration occurring at greater portions of the total local watershed, up to 100% of the subcatchment if the historical conditions at each location were classified as wetlands. In sum, we explored restoration outcomes by generating scenarios at two different scales: 1) Ruamahanga Basin scale (n = 19 scenarios) and 2) multiple subcatchments (n = 302 scenarios) nested within the Ruamahanga Basin (Table 1). Subcatchments were delineated using the areas contributing surface-flow to 402 uniquely identified historical wetlands.

Table 1
Differences between the Ruamahanga Basin and test-catchment scale approaches for exploring wetland restoration and ecosystem services.

	Spatial scale of scenarios	
	Ruamahanga Basin	Subcatchments nested within the Ruamahanga Basin
Number of iterations/ scenarios	(n = 19) Incremental increases in the percentage of the historical wetland extent restored until full restoration is reached	(n = 302) Sub catchments > 1 km^2 each run as both restored and unrestored (302 × 2) to capture gains achieved from restoration
% of historical wetland restored	0, 5, 10, 15 …95 %	100% (within the subcatchment)
% of catchment area restored as wetland	0, 1.33, 2.67, …-26 % (of basin)	<1–100% (of subcatchment)
Watershed position of restoration	Random placement within each historical wetland	Catchment outlet, 100% of the wetland's historical extent
Likely decision maker	Regional council	Landowner or catchment group
Decision context/ question	How much wetland restoration is required to achieve ecosystem service targets?	What is the change in ecosystem services from restoring this (one) wetland?

2.3. *Ruamahanga basin scale (19 Scenarios) -Restoring from 0 to 95% of historical wetlands*

The Ruamahanga Basin scale allowed us to explore consequences of different restoration targets from the perspective of broad-scale decision makers, such as regional councils. Random locations in each and every historical wetland (n = 402) were selected on the map using ESRI's ArcGIS (Fig. 2). Restoration was initiated at these random points, and using a spread tool, restoration was extended from these points in eight

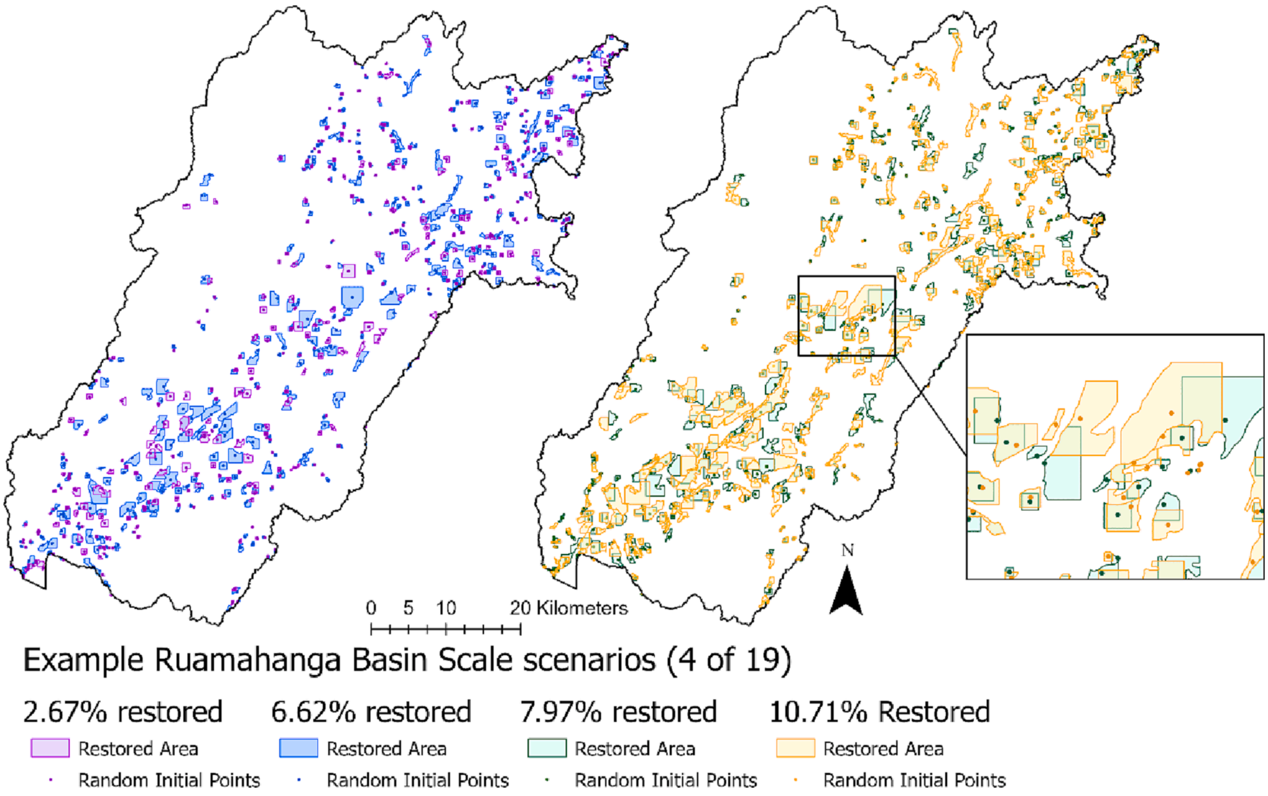


Fig. 2. Different initiation points led to different restored areas for each scenario. Here we show 4 of 19 of the Ruamahanga Basin scale scenarios). Wetlands are restored in random locations (Random Initial Points, n = 402) within the historical wetland extent, and then from these random points, which differ by scenario, restoration is extended to fill a target extent (e.g., 2.67 % of the basin).

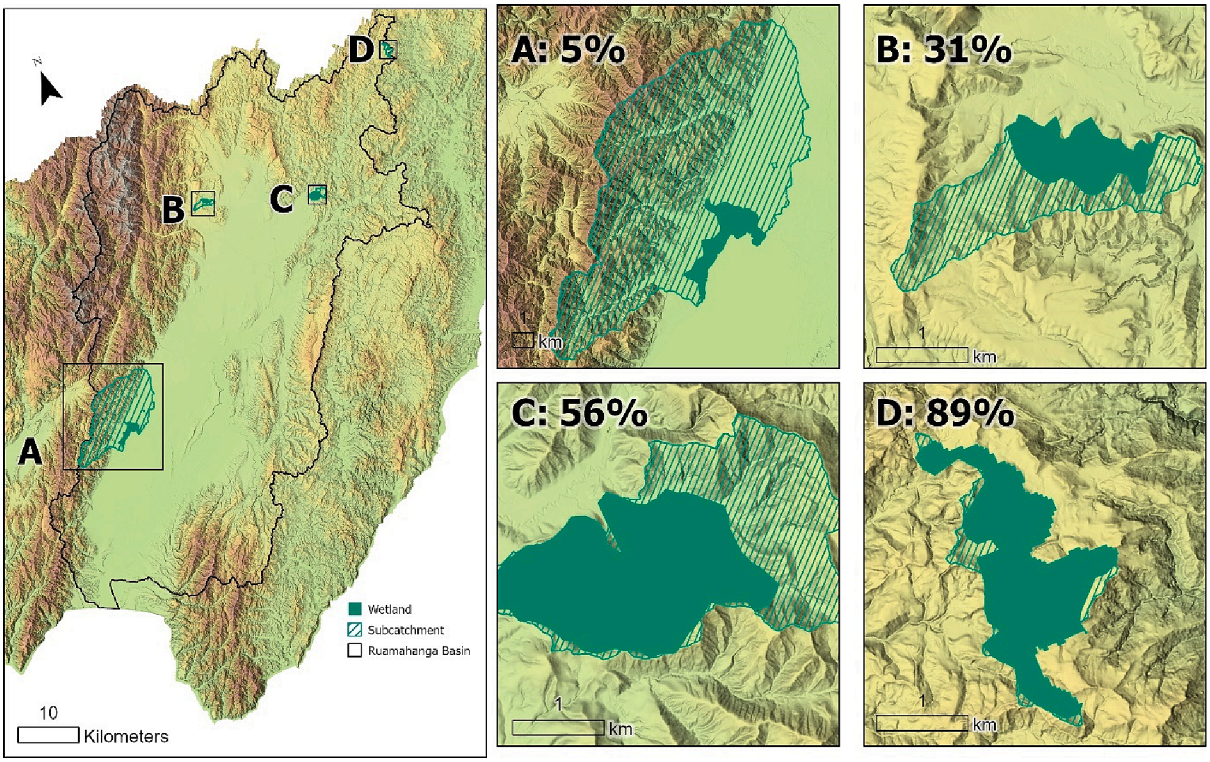


Fig. 3. Examples (4 of 302) of restored wetlands (filled green) and their contributing areas (hashed green). The wetland used to delineate the subcatchment comprises a different percentage of the subcatchment ranging from <1% to 100% of the subcatchment Here we show four examples found throughout the Ruamahanga basin (5%, 31%, 56%, and 89%). We defined the 302 subcatchments by delineating contributing areas to historical wetlands within the Ruamahanga basin. See Fig. 4 for maps of the subcatchments.

compass directions until the desired area was reached (Chubaty and McIntire, 2020). We restored individual wetlands in increments of ~five percent, resulting in nineteen scenarios (0–95% of total historical wetland extent restored). We created a new set of random points for each of the 18 restoration scenarios (1 scenario was unrestored). Restoring in random locations reflects current opportunistic practices, which do not target specific positions on the landscape.

2.4. Subcatchment scale -restoring from 0 to 100% of each subcatchment to wetland

Our subcatchment scale approaches restoration from the perspective of a decision maker restoring an individual wetland (Fig. 3, Fig. 4). Areas contributing surface-runoff water for each historical wetland were mapped using ArcGIS hydrology tools. Wetlands with contributing areas of <1 km^2 were excluded from the study to ensure the contributing areas were not a result of DEM artefacts. This exclusion of small contributing areas (n = 100) resulted in 302 subcatchments, ranging in size from ~1 km^2 to 200 km^2 (Fig. 4). Next, we determined how much of these contributing areas were historically wetland, which ranged from <1% to nearly 100%. These highly varying wetland extents allowed us to infer how much wetland restoration as a percentage of the total catchment area is needed to affect measurable ecosystem service gains. The size of the contributing area was significantly, albeit weakly, correlated with the percentage of historical wetland (Pearson's r ~ 0.30, $p < 0.01$), justifying going forward with our subcatchment analysis. One benefit of this approach is that all the wetland restoration took place in the same position within the catchment-surrounding the outlet, which allowed us to control for landscape position, which can be an important determinant of ecosystem service outcomes (Tran et al., 2022). One drawback of our subcatchment approach is that there may be fundamental biophysical differences (e.g., terrain/climate/soil) among subcatchments with different proportions of wetland, which could influence the provisioning of ecosystem services but are unaccounted for in our approach.

2.5. Ecosystem service models

2.5.1. LUCI modelling

Because wetlands are often small, high-resolution models are needed to quantify the ecosystem services they provide (Tomscha et al., 2019). We also know that small wetlands can be disproportionately important for certain ecosystem services, such as nutrient retention (Cheng and Basu, 2017). Our ecosystem services modelling was done through the modelling tool Land Use Capability Indicatory (LUCI) (https://www.lucitools.org/). Formerly called Polyscape, LUCI is an ecosystem service modelling framework, which provides freely available models for multiple ecosystem services, biodiversity, and their trade-offs. LUCI runs on ESRI's ArcGIS (Jackson et al., 2013; Trodahl et al., 2017b). LUCI is a suitable tool for modelling changes in ecosystem services at spatial resolutions of 5 m^2, providing an ideal tool for modelling ecosystem services associated with wetland restoration. Our simulations focus on biophysical mechanisms (Häyhä and Franzese, 2014), which LUCI is well-suited to model due to its high-resolution nutrient modelling tools, ability to model across different scales, as well as prior validation in a New Zealand environment (Bagstad et al., 2013). LUCI requires several GIS layers as input, including a digital elevation model, land cover, soil, and climate data. We used freely available spatial data in Aotearoa New Zealand, with minor amendments (Table 2). LUCI tools were run on both the restored and unrestored landscapes for a total of 19 runs at the Ruamahanga Basin scale and 604 subcatchment runs, each at a

Table 2
Summary of datasets used in LUCI to estimate ecosystem service gains from incremental increases in restored wetland extent. Descriptions ().

Dataset	Description	Source
Digital elevation model (DEM)	1 m LIDAR point cloud resampled to a 5 m raster	Greater Wellington Regional Council (LINZ, 2013)
Soil	Fundamental Soils Layer, vector polygons based on soil surveys (1930′s-present), resolution variable	(Landcare Research, 2015)
Land cover	New Zealand Land Cover database LCDB (v5), 33 landcover types across New Zealand identified largely using SPOT imagery	(Landcare Research, 2020)
Precipitation	Average annual rainfall in mm yr^{-1} at a 0.5 km grid resolution	(NIWA, 2018)
Evapotranspiration	0.5 km grid resolution	(NIWA, 2018)

adapted from Tomscha et al., 2019

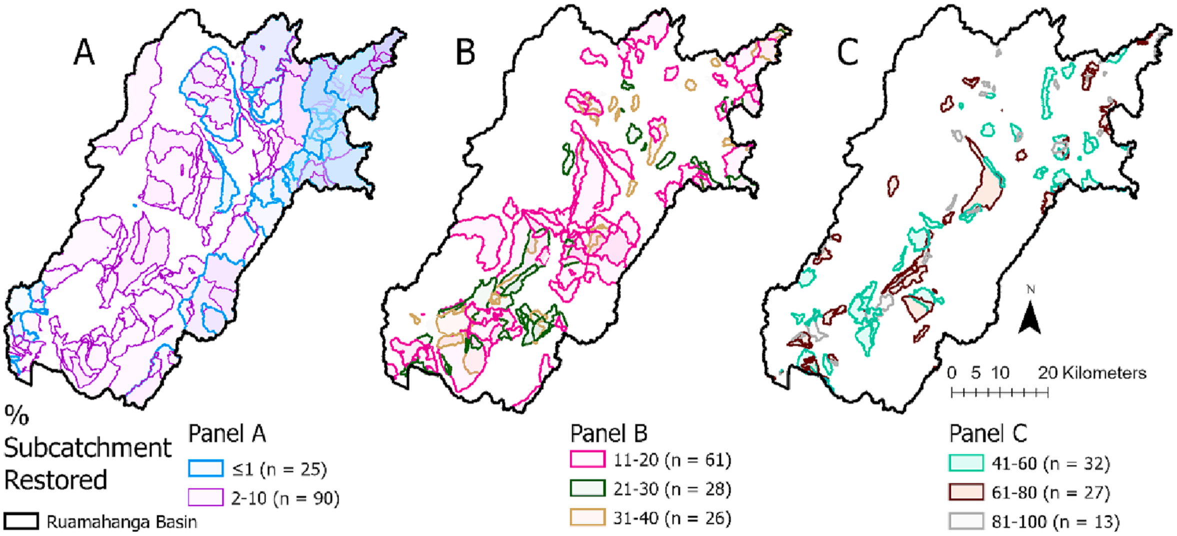


Fig. 4. Individual subcatchments (i.e. contributing areas) of different historical wetlands (302 of 302), which overlap and vary greatly in size. Historical wetlands extended across <1% to 100% of these subcatchments, Panel A shows 115 subcatchments where wetlands comprise a lower (<11%) percentage of the catchment area, average subcatchment size = 663 km^2. Panel B shows 115 subcatchments where wetlands comprise a medium (11–40%) percentage the wetland area, average subcatchment size = 143 km^2. Panel C shows 72 subcatchments were wetlands comprise a high (41%–100%) percentage of the subcatchment area, average subcatchment size = 57 km^2.

resolution of 5 m × 5 m pixels (Table 1), assessing four ecosystem services in relation to recognised thresholds (Table 3).

2.5.2. *Agriculture*

Agricultural productivity was modelled based on a combination of land cover maps, aspect, slope, soil fertility, and drainage (The LUCI team, 2019). Each raster is classified based on a categorical ranking of its potential agricultural productivity. For example, to be considered highly productive, the maximum productivity value, the slope must be <5% and elevation must be lower than 350 m, and soil fertility must be high. Then this potential productivity is contrasted to the raster cell's current use. After assessing potential agricultural productivity across a landscape, LUCI evaluates whether the current land use is a good fit for its potential productivity. For example, if the raster cell is potentially very productive, but currently a forest, the cell is considered negligibly productive. The output is divided into four ranked categories: High productivity, Moderate productivity, Marginal productivity, and Negligible productivity. To explore change in this ecosystem service, we describe the total area that switches category following restoration. We note that any loss of productivity is likely to be undesirable to farmers, and from a societal point-of-view loss of highly productive land to restoration is likely to be the most contentious.

2.5.3. *Carbon*

The IPCC Tier 2 protocol was used to evaluate carbon storage (The LUCI team, 2019). For the Tier 2 protocol, carbon storage estimates are based on a combination of soil and land cover data, which have been parameterized using Aotearoa New Zealand-specific data linking carbon estimates to New Zealand Land Cover Database LCDB (v5). We report carbon storage in tonnes per hectare, and subsequently, convert these to annual emissions per person to provide a tangible comparison for how wetland restoration can contribute to emissions targets. The threshold we adopted for carbon storage is the amount that offsets the per capita carbon emissions of the population of the Wairarapa (48,480 individuals in 2018), the administrative area in which the Ruamahanga Basin lies (Stats NZ, 2018). For per capita carbon emissions, we used the average annual emissions of 7.5 tonnes per capita in 2018 (Ministry for the Environment, 2021) (https://environment.govt.nz/publications/new-zealands-greenhouse-gas-inventory-1990–2019-snapshot/how-new-zealand-compares-to-other-countries).

Table 3
Overview of the policy-relevant thresholds and unit of measurements for ecosystem services at two scales (Ruamahanga Basin and sub catchment).

Ecosystem service	Scale		Major assumptions and limitations of the threshold
	Ruamahanga Basin	Subcatchments	
Nitrogen retention	Thresholds based on risk of colorectal cancers >0.87 mg/L		• Total N = Nitrates Not spatially explicit Doesn't consider ecological thresholds
Phosphorus retention	Thresholds based on boundaries between oligotrophic, mesotrophic and eutrophic streams >0.025 mg/L (moderate concentration) >0.075 mg/L (high concentration)		• Not spatially explicit for flowing water
Carbon Storage	Δ tonnes C Thresholds based on offsetting population of the Wairarapa	None	• New Zealand average emissions apply to the Wairarapa,7.5 tonnes per capita (2018 estimate) (Ministry for the Environment 2021) Arbitrary population boundary
Agricultural production	Δ ha of land productivity categories Land is categorized into four classes: -Highly productive -Moderately productive -Marginally productive -Negligibly productive No thresholds specified		• Productive value of land is equivalent to profit and goals for farming

2.5.4. *Nitrogen and phosphorous modelling*

Nitrogen and phosphorus retention were modelled in LUCI using a modified export coefficient approach, export coefficients being a long-standing approach in modelling nutrient flows (White et al., 2015). Topographic routing of water flow combined with data on rainfall and information on N and P retention and export per land cover type provided an estimate of the nutrient load present at any given grid cell. Both dissolved and particulate pathways were calculated and summed to obtained total export and retention values. Stream concentrations of both N and P were estimated along the stream network based on these terrestrial flows, providing an annual average concentration of both N and P (The LUCI team, 2019). Point sources can be considered, but the process of collating data on wastewater discharges, sewage tanks etc. was prohibitive and hence only diffuse pollution sources were considered. We validated the model for our unrestored scenario (see Appendix 1), and additional validation of these models in a New Zealand context are demonstrated elsewhere (Taylor, 2018; Trodahl et al., 2017a).

2.5.5. *Nitrogen thresholds*

The impacts of high nitrogen levels in waterways include both ecosystem state changes, such as eutrophication and excess periphyton growth, as well as human health implications, such as increases in colorectal cancers (Ministry for the Environment, 2020; Richards et al., 2021). Hence, setting limits for nitrogen is multifaceted. Here, we use a simple standardized approach focusing on the impacts to human health, exceedance of the 0.87 mg NO_3.N/L recommended for reducing colorectal cancers in the restored and unrestored wetland scenarios (Richards et al., 2021). Our approach is conservative, because we know not all N in streams will be nitrates, however, in the Aotearoa New Zealand, where survivorship of colorectal cancers is lower in Māori and Pasifika communities, a cautious approach to nitrogen levels is warranted (Sharples et al., 2018). Our approach is also conservative relative to the World Health Organization recommended threshold of 11.3 mg/L. LUCI models total N but we assume, in the context of the Ruamahanga Basin, that this is mostly nitrate because it is more mobile in soils than other forms of N (Paul and Clark, 1996) and is therefore likely to leach to waterways in agricultural contexts (Di and Cameron, 2002). Furthermore, evidence suggests nitrate is more abundant than other forms of nitrogen within a local wetland within the Ruamahanga Basin (Waring, 2017).

2.5.6. *Phosphorus thresholds*

Thresholds for total P are established by the New Zealand Ministry for the Environment and are often spatially explicit (Ministry for the Environment, 2020). As a general guideline, total P thresholds of 0.025 and 0.075 mg/L are suggested as the boundaries between oligotrophic and mesotrophic and the boundary between mesotrophic and eutrophic streams, based on guidance from the literature (Dodds et al., 1998; The LUCI team, 2019). Phosphorous is problematic in the Wairarapa more generally, as Lake Wairarapa's phosphorus concentration classifies it as supertrophic (Cockeram et al., 2014). However, there is some evidence that phosphorus may be decreasing in NZ waterways (McDowell et al., 2019).

3. Results

3.1. Basin-scale restoration

3.1.1. Nitrogen and phosphorus retention for improved water quality

Basin-scale restoration improved N and P in-stream concentrations, but high soil N concentrations remained widespread. Most (60.1%) of the Ruamahanga basin is modelled to be affected by soil–water concentrations of nitrogen that may increase the risk of colorectal cancers, if consumed (Fig. 5). With restoration of 95% of historical wetlands, which amounts to ~688 km^2 of restored area, this percentage falls to 40.7 %. We found a linear relationship between the area set aside for wetland restoration and the gains in nitrogen retention. Phosphorus concentrations higher than 0.025 mg/L were predicted to be found across more than 40% of the basin, of which 17.3% exceed 0.075 mg/L. Restoring 95% of historical wetlands, is predicted to almost halve the number of locations that exceed 0.075 mg/L to 9.1% of the total. We found a linear relationship between the area set aside for wetland restoration and the gains in phosphorus retention.

3.1.2. What co-occurring gains are achieved for carbon?

Carbon gains are incremental according to our IPCC tier 2 models, reaching gains of 600,000 tonnes of carbon with 95% of the historical wetland restored. Just over 45% of the historical wetland in the Ruamahanga Basin would need to be restored to offset the annual carbon emissions of the Wairarapa. However, offsetting the carbon equivalent of all gases per capita (16.9 tonnes per capita), is not possible through wetland restoration, which requires gains of 819,312 tonnes (MfE, 2021).

3.1.3. How much agricultural land is lost in each scenario?

Modelled wetland restoration, for the most part, took highly productive land out of production. While most historical wetland locations are considered highly productive, a small percentage of the land is considered moderately and marginally productive. In our 95% restoration scenario, 8.27 km^2 were initially moderately productive, and 2.27 km^2 were initially marginally productive. These locations represent obvious opportunities for restoration without sacrificing highly productive agricultural lands.

3.2. Subcatchment restoration

3.2.1. How much restoration can achieve water quality standards for N and P?

Restoration helped most subcatchments achieve targets for water quality across a range of percent restored (Fig. 6). At ~60% of a total catchment restored, all but one basin achieved N levels below 0.87 mg/L, which would ensure N in these waterways would not contribute to colorectal cancer risk. Although it should be noted that increases in colorectal cancers are incremental with nitrate levels, which means any reduction in nitrates would be positive towards the goal of reducing rates of colorectal cancers. For phosphorus, reductions were poorly linked to percent subcatchment restored. Nonetheless, wetland restoration reduced P levels below thresholds. Here, in all cases, restoration reduced P levels at the stream outlet to levels below 0.025 mg/L (Fig. 6).

3.2.2. Carbon sequestration gains and losses in agricultural production

Sequestered carbon primarily increased through wetland restoration, but there is no clear trend in how this relates to percent subcatchment restored as wetland, increases in carbon were highly variable but slighter greater when >50% of the subcatchment was restored (Fig. 7). On the other hand, agricultural production primarily decreases with wetland restoration, but again with no obvious relationship with percent of subcatchment restored (Fig. 7).

3.2.3. Do gains and losses accelerate, decelerate or change proportionately with percentage area restored?

At the basin scale, all ecosystem services change proportionately and linearly with restored area (Fig. 4). However, at the subcatchment scale, changes for N and P were nonlinear. We found rapid gains in N and P removal between 0 and 30% followed by an attenuation of gains from ~30–70% of subcatchment restoration (Fig. 7). This attenuation was greater for P than for N. For both N and P, gains accelerated again above ~60% of subcatchment restoration. There was no clear pattern for changes in carbon and agricultural productivity at the subcatchment scale.

4. Discussion

Our main finding from restoring historical wetlands incrementally and randomly across the Ruamahanga Basin scale is that we found no percentage at which restoration did not add benefits for N and P retention and carbon storage, and no percentage at which restoration did not detract from agriculture. At the basin scale, overall gains and losses in ecosystem services were incremental. However, at the subcatchment scale, more variability emerged for water quality. While gains in carbon storage and loss in agricultural productivity are unsurprisingly related to the total area restored, rather than the percentage of subcatchment restored, gains in N retention are closely related to the percentage of subcatchment restored, with nearly all subcatchment water nitrate levels meeting threshold targets when greater than 60% of the subcatchment is restored. Below, we discuss the implications of these findings at different scales.

At the subcatchment scale, most catchments restored at >60% achieved compliance for nitrogen targets and phosphorus, and as such, 60% restored holds promise as a target for achieving N and P standards for specific locations where these concentrations are problematic. At the basin scale, however, problematic levels of nitrates remained spatially extensive. These spatially extensive high N concentrations represent soil water, and are therefore unlikely to be consumed *in situ*, but may have long-term impacts on groundwater nitrate levels (Basu et al., 2022; Rajanayaka et al., 2020). In other words, the implications of extensive high N concentrations in soils are likely to emerge only in the future as high groundwater nitrogen. To mitigate N across the Ruamahanga Basin, restoration of upland ecosystems would also be needed.

Carbon emissions for the population of the region were offset for one year by restoring ~13% of the Ruamahanga Basin. Here, we assumed full restoration to indigenous forest cover, but forest restoration is clearly not instantaneous, as the accumulation of carbon takes place over decades to centuries (Carswell et al., 2012). Furthermore, while swamp forests were formerly extensive, not all wetland types in the study region support trees (Tomscha et al. 2019), calling into question the plausibility of forest cover in all wetland sites. Nonetheless, our analyses make clear the vast amount of restoration needed to offset only one year of carbon emissions, especially when restoring to indigenous forest. Carbon credit plantings in Aotearoa New Zealand, which do not credit biodiversity, currently incentivise planting exotic pine and eucalypt species, which would achieve higher carbon credits than indigenous forest. The benefit of our method for mapping carbon storage is that it aligns with how emission-trading schemes credit land cover changes for carbon. However, we know carbon storage is considerably more variable than shown by Tier 2 approaches. One limitation with Tier 2 carbon storage methods is the lack of heterogeneity in reported carbon storage, which remains constant across land cover types (Buendia et al., 2019; Smith et al., 2012).

Losses in agricultural production were incremental across the basin-scale scenario and showed no clear pattern at the subcatchment scale. However, we note that some historical wetlands are classified as moderately or marginally productive currently. Landowners may be more willing to restore wetlands on marginally productive land, and thus, locating these parcels is key for expanding the spatial extent of

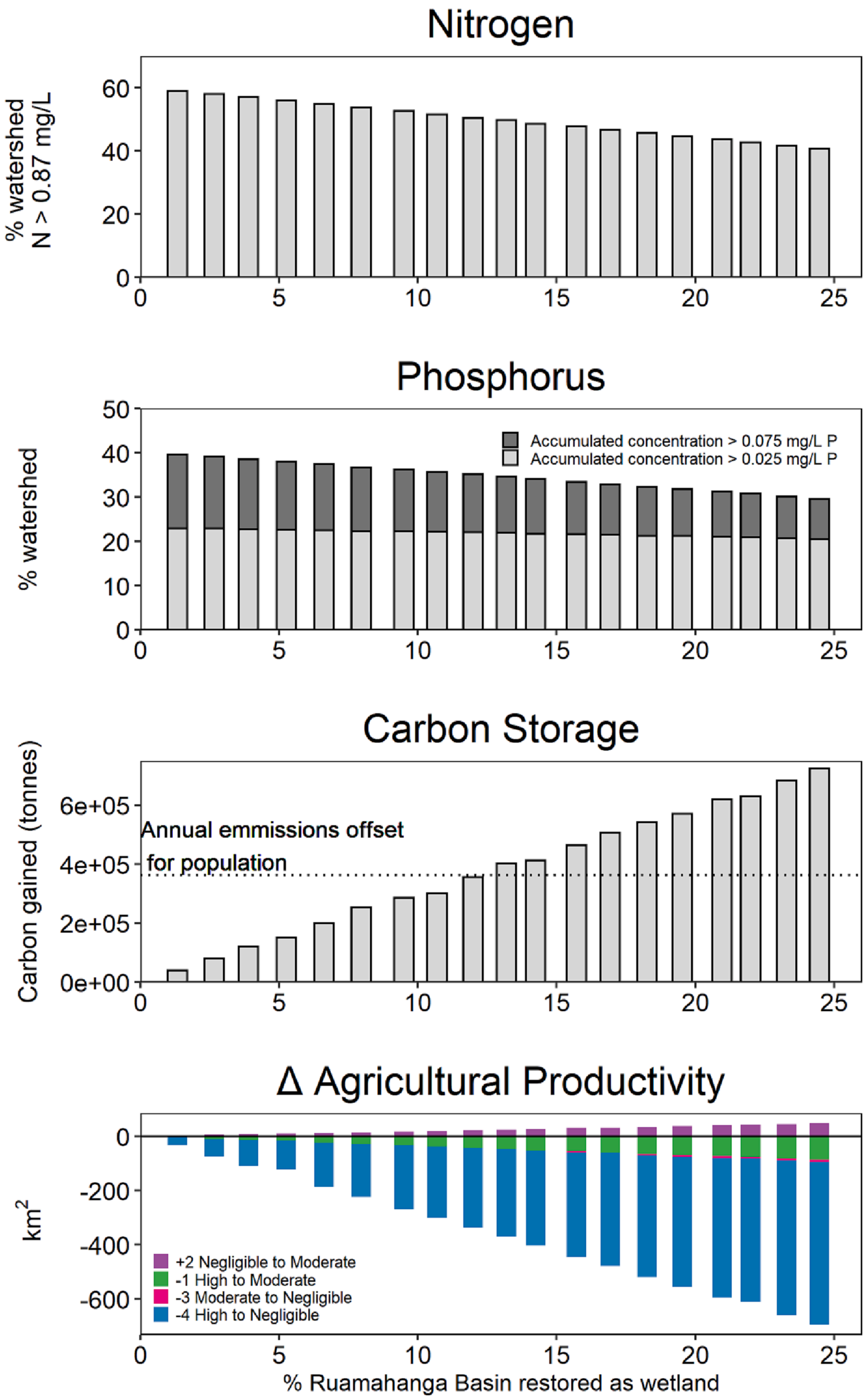


Fig. 5. Changes in ecosystem services (Nitrogen retention, Phosphorus retention, Carbon storage, and Agricultural productivity) following restoration of different percentages of historical wetlands comprising different percentages of the entire Ruamahanga Basin (x axis). Dotted line shows how much needs to be restored to offset the emissions for population of the Wairarapa for one year. Just over 12% of the basin (45% of the historical wetland extent) needs to be restored to account for the population's emissions for one year. For agricultural productivity changes from Marginal to Negligible, Marginal to Moderate, and Very marginal to Negligible productivity are not shown because they comprise <0.005 km^2.

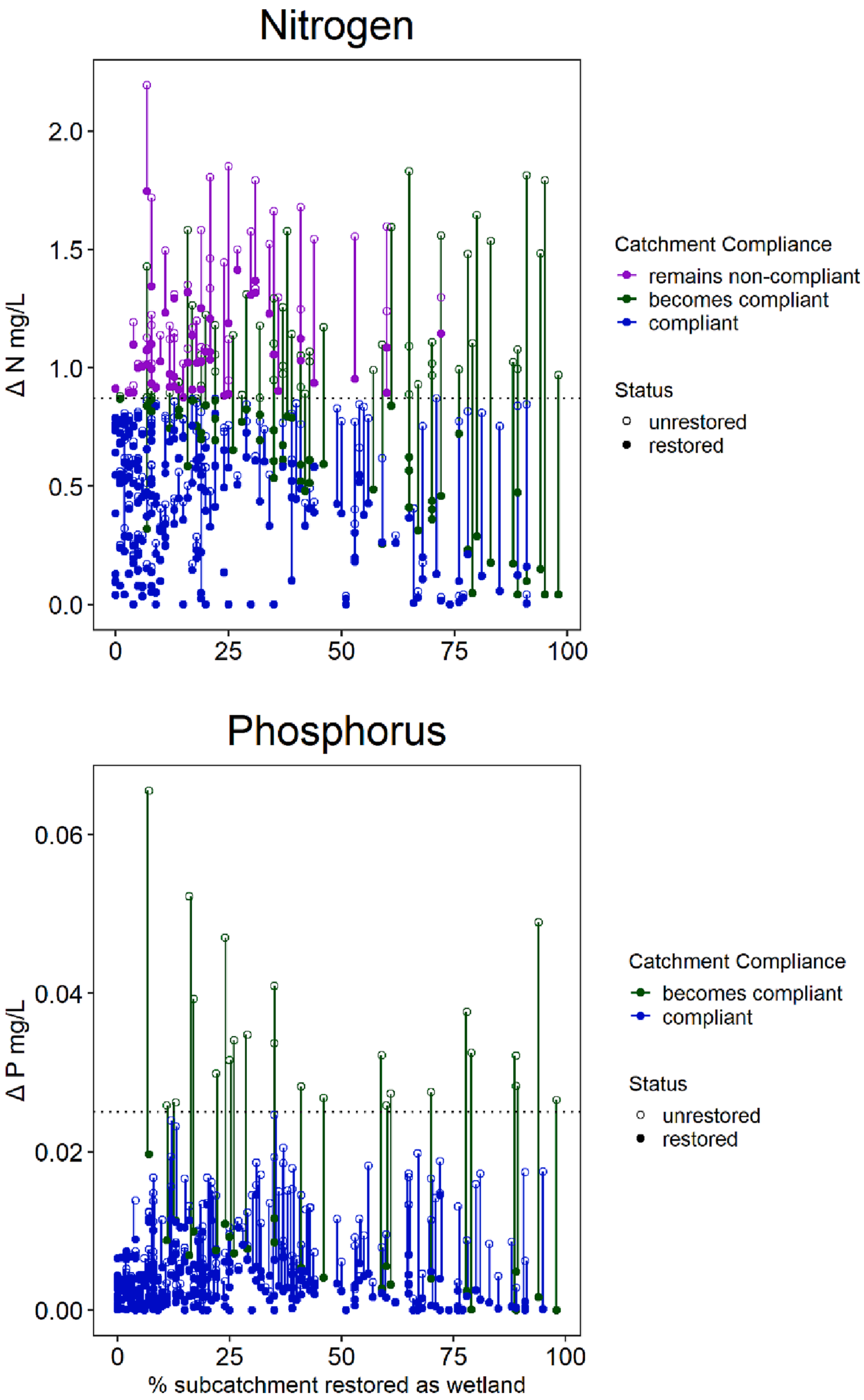


Fig. 6. N and P levels for unrestored (unfilled dots) and restored (filled dots) subcatchment areas. Each subcatchment has two data points in each graph, one unfilled dot showing nutrient concentration at the outlet of the subcatchment with its wetland unrestored, and one filled dot showing the nutrient concentration at the outlet of the subcatchment following wetland restoration. These two points are adjoined by a line that shows the magnitude of difference following restoration. The x axis shows the level of restoration for each subcatchment which ranges from 0 to 100%. The dashed line indicates the target level for these nutrients. (A) After subcatchments restored to ~60% wetland, only one catchment exceeded the N compliance threshold of 0.87 mg/L. (B) Restoration was effective at reducing P across all percentages of restoration.

restoration. Nonetheless, acceptable reductions in agricultural production are rarely explicit in conversations about trade-offs between agricultural production and other ecosystem services, in part, because productivity matters at the farm/individual-scale and reflects management and profitability decisions of farmers. Tolerable reduction in profits or production may vary with the size and type of farm and may be best incentivized via agri-environmental schemes (Jellinek et al., 2013).

A global increase in the human population means that more food production may be needed for future generations. While total calories currently produced are sufficient to meet global needs, nearly 800 million and 2 billion people suffer from under-nourishment or micronutrient deficiencies, respectively (Ramankutty et al., 2018; Theodore, 2010). Despite being challenging, managing trade-offs between agricultural production and other ecosystem services is necessary to reduce carbon emissions and in-stream pollution caused by agriculture. Importantly, reducing carbon emissions also matters for sustained food

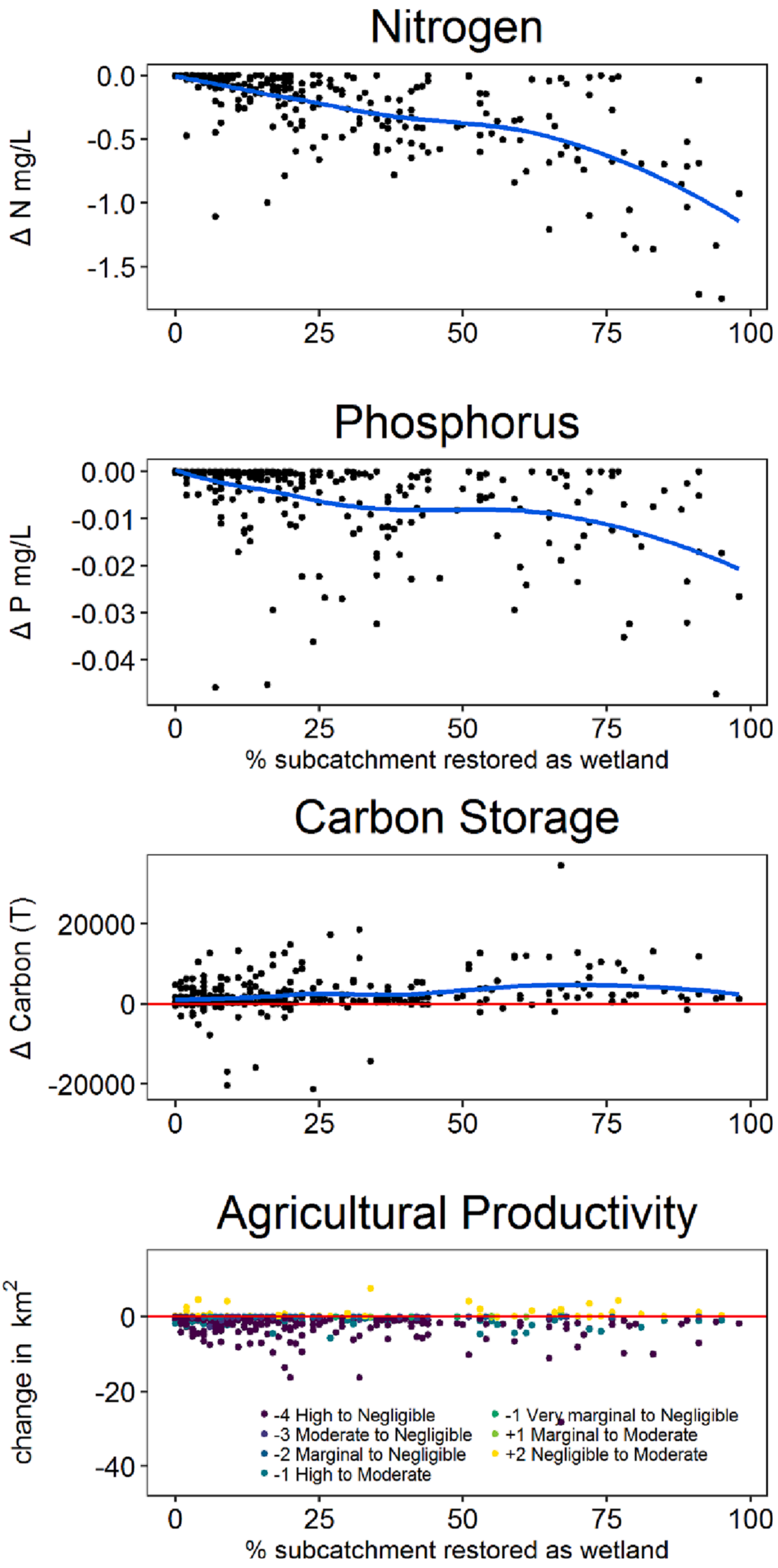


Fig. 7. Change in nitrogen, phosphorus, carbon storage and agricultural productivity as a function of percent of catchment restored as wetland. Horizontal red lines indicate 0 on the y axis.

production because such emissions feedback and reduce the productivity of future agriculture, having already reduced production since the 1980's (Lobell et al., 2011). Finally, wetlands support fisheries through providing fish nursery habitat and by improving water quality (Diaz and Rosenberg, 2008), and as such, the loss of productive land should be balanced against increased production in fisheries.

4.1. *Uncertainty and thresholds in target setting*

Understanding whether social-ecological change is incremental or non-linear is an important area of research in the Anthropocene, especially as it relates to human wellbeing and the delivery of ecosystem services. One widely accepted conceptual model of is that ecosystems can exhibit multiple stable states, and changes between these states can be driven by external pressures. Thresholds determine when ecosystems will switch states. Thresholds are appealing to policy makers, because if a threshold exists, then clear targets can be set to ensure a safe operating space for ecosystem services. However, there isn't always evidence that threshold models are appropriate (Hillebrand et al., 2020; Qian and Cuffney, 2012). Furthermore, there is even less evidence that these thresholds hold at multiple spatial scales for ecosystem services (Spake et al., 2022). In some cases, a continuous change model is more applicable (Banks-Leite et al., 2021). Here we show that at the basin scale, a continuous change model applied to N and P retention, while at the subcatchment scale, threshold dynamics were more likely at play. We also found a great deal of variability in results. For example, some of our subcatchments achieved large gains towards our water quality targets, despite low percentages of restoration. As such, applying a consistent percentage restored target may be overly costly in these cases. Understanding why these responses vary would be key to creating more targeted policies.

One important point to note is that the amount of restoration needed will vary greatly depending on ecosystem service targets set by policy makers. Setting targets for each ecosystem service involves exploring the trade-offs, as well as the risks, of setting targets either too low or too high. For example, setting N limits too high may mean a stream is at increased risk of switching to a eutrophic state (Snelder et al., 2019). Here, we show the percent restoration needed to reach several relevant targets for water quality and carbon, but these ecosystem service targets remain debated in policy realms. For example, the World Health Organization suggests nitrate levels be below 11.3 mg/L for drinking water, but there is evidence that incidence of colorectal cancers increase above 0.87 mg/L (Richards et al., 2021). These targets also differ if the underlying goal is based on ecosystem state rather than human health. The N targets set in this paper are conservative relative to targets set to limit periphyton growth, which are the primary targets underpinning many of Aotearoa New Zealand's policies for freshwater (Snelder et al., 2019). These periphyton thresholds vary based on characteristics of individual rivers, as well as the substrate type of the river. A deeper exploration of the consequences of different threshold targets is warranted, and the consequences of exceeding a range of thresholds in different locations would provide better context for these results.

Another challenge to setting targets is that production and delivery of ecosystem services may occur at different scales. For example, climate regulation is a global scale ecosystem service, while water quality benefits are delivered locally, but based on upstream spatial dependencies (Costanza, 2008). These differences in scale create boundary issues when determining whether a particular target for restoration has been reached. For example, do the people living within a particular geographical area need to offset their carbon emissions locally, or can carbon storage be better achieved elsewhere? How can we distribute responsibility for upstream polluters for downstream water quality? These questions and challenges necessitate robust networks for cross-scale communication.

4.2. *Cross-scale knowledge gaps*

Ecosystem service assessments are commonly conducted across large regions or basins (Dang et al., 2021). However, we show here that much of the variation in outcomes of ecosystem services happens at scales below the basin scale. This variation results in localized ecosystem service effects and shows that there are benefits in considering the perspectives of community groups or individual land holders, as well as across multiple scales, when exploring ecosystem service outcomes (Scholes et al., 2013). Furthermore, at the local scale, i.e., from the perspective of individuals, knowledge gaps in ecosystem services are common. These information gaps impede restoration practitioners and regional managers from making decisions that are effective across scales. For example, at the local subcatchment scale for wetlands, most individuals would not have information on the boundaries of their wetland's contributing area, which means information on the relative percent of subcatchment restored would not be known. Nor do farmers usually know the portion of in-stream nutrients that can be attributed to their farm. Likewise, when looking across the catchment, managers may have poor knowledge of the management of local fields or paddocks. For example, tile drains may not be adequately documented and thus the land may be more productive than our modelling suggests. Maps, such as those produced in these analyses, help to fill these knowledge gaps.

Simple targets for protection, such as 30% by 2030 for enhancing biodiversity, may be appealing. However, from an ecosystem service perspective, apart from N and P retention at the local scale, we show no evidence there is a point at which gains in ecosystem services taper off with an increasing percentage of land restored. Furthermore, variable outcomes shown in subcatchment wetland restoration demonstrate the results of simple percent-area targets may be heterogenous across decision-making scales. A highly targeted and optimised restoration of 10% of a basin may have greater benefits than random or opportunistic restoration of 30% of the land. Clarification about how targets of can be achieved for different ecosystem services may facilitate better implementation at local and catchment scales. For some ecosystem services, restoration of more than 30% of land area may be required. Nonetheless, percent-area targets retain some value. Perhaps one of the greatest strengths of percentage-area restoration targets is that they communicate ambition and commitment to a goal. Ambitious targets are needed to improve biodiversity (Díaz et al., 2020). Aotearoa New Zealand has success in protecting more than 30 percent of its land area, and yet still faces threats to species conservation and water quality declines.

5. Conclusion

We explored how restoring an increasing percentage of a catchment at two scales may help achieve targets for water quality and carbon storage. We found that at the basin scale, gains in water quality and carbon storage were incremental and more-or-less linearly related to the percentage of area restored. There was no point at which gains in N and P attenuated until the historical wetland extent was fully restored. However, results at the subcatchment scale were highly variable, showing a non-linear relationship with increasing percentages of restoration. Proportionate gains in N and P occurred most rapidly between zero and 30% restoration and again above 60% restoration. These differences in outcomes at different scales reflect the importance of identifying the appropriate scale for ecosystem service assessments and management interventions. The importance of conducting smaller scale assessments to understand ecosystem service variability and outcomes for specific locations, has likely been underappreciated in the past. While Aotearoa New Zealand has already reached the UN's proposed target of 30% by 2030 by protecting large areas of mountainous terrain, losses in species, lowland habitat types and declines in ecosystem services remain a concern. Restoring wetlands, a highly underrepresented ecosystem in Aotearoa New Zealand, holds promise for improving multiple ecosystem services.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

This research was funded by the Holdsworth Charitable Trust and Bioprotection Aotearoa. We thank reviewers for their comments on earlier versions of this manuscript. This research took place on the traditional territory of Ngāti Kahungunu ki Wairarapa and Rangitāne o Wairarapa.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2023.101527.

References

Arroyo-Rodríguez, V., Fahrig, L., Tabarelli, M., Watling, J.I., Tischendorf, L., Benchimol, M., Cazetta, E., Faria, D., Leal, I.R., Melo, F.P.L., Morante-Filho, J.C., Santos, B.A., Arasa-Gisbert, R., Arce-Peña, N., Cervantes-López, M.J., Cudney-Valenzuela, S., Galán-Acedo, C., San-José, M., Vieira, I.C.G., Slik, J.W.F., Nowakowski, A.J., Tscharntke, T., 2020. Designing optimal human-modified landscapes for forest biodiversity conservation. Ecol. Lett. 23, 1404–1420. https://doi.org/10.1111/ele.13535.

Ausseil, A.-G., Gerbeaux, P., Chadderton, L.W., Stephens, T., Brown, D., Leathwick, J., 2008. Wetland ecosystems of national importance for biodiversity: criteria, methods, and candidate list of nationally important inland waters.

Bagstad, K.J., Semmens, D.J., Waage, S., Winthrop, R., 2013. A comparative assessment of decision-support tools for ecosystem services quantification and valuation. Ecosyst. Serv. 5, 27–39. https://doi.org/10.1016/j.ecoser.2013.07.004.

Banks-Leite, C., Larrosa, C., Carrasco, L.R., Milner-Gulland, E.J., 2021. The suggestion that landscapes should contain 40 of forest cover lacks evidence and is problematic. Ecol. Lett. 24, 1112–1113.

Basu, N.B., van Meter, K.J., Byrnes, D.K., van Cappellen, P., Brouwer, R., Jacobsen, B.H., Jarsjö, J., Rudolph, D.L., Cunha, M.C., Nelson, N., Bhattacharya, R., Destouni, G., Olsen, S.B., 2022. Managing nitrogen legacies to accelerate water quality improvement. Nat. Geosci. 15 https://doi.org/10.1038/s41561-021-00889-9.

Bentley, S., Tomscha, S.A., Deslippe, J.R., 2022. Indicators of wetland health improve following small-scale ecological restoration on private land. Sci. Total Environ.

Buendia, E.C., Guendehou, S., Limmeechokchai, B., Pipatti, R., Rojas, Y., Sturgiss, R., Tanabe, K., Wirth, T., Romano, D., Witi, J., Garg, A., Weitz, M.M., Cai, B., Deborah A. Ottinger, H.D., MacDonald, J.D., Ogle, S.M., Rocha, M.T., Sanchez, M.J.S., Bartram, D.M., Towprayoon, S., 2019. Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories.

Carroll, C., Noss, R.F., 2021. How percentage-protected targets can support positive biodiversity outcomes. Conserv. Biol. 1–10 https://doi.org/10.1111/cobi.13869.

Carswell, F.E., Burrows, L.E., Hall, G.M.J., Mason, N.W.H., Allen, R.B., 2012. Carbon and plant diversity gain during 200 years of woody succession in lowland New Zealand. N. Z. J. Ecol. 36, 191–202.

CBD, 2020. CBD - Global Biodiversity Outlook 5 - FULL REPORT.

Cheng, F., Basu, N.B., 2017. Landscape nutrient processing. Water Resour. Res. 53, 1–19. https://doi.org/10.1002/2016WR020102.Received.

Chubaty, A.M., McIntire, E.J., 2020. SpaDES.tools: Additional Tools for Developing Spatially Explicit Discrete Event Simulation (SpaDES) Models. R package version 0.3.6.

Clarkson, B.R., Ausseil, A.E., Gerbeaux, P., 2013. Wetland Ecosystem Services.

Cockeram, B., Perrie, A., Council, G.W.R., 2014. Lakes State of the Environment monitoring programme: Annual data report, 2013/14.

Costanza, R., 2008. Ecosystem services: Multiple classification systems are needed. Biol. Conserv. 141, 350–352. https://doi.org/10.1007/bf01844766.

Crisp, P., 2020. Priority Biodiversity Conservation Actions for the Wellington Region – A Think-piece. Wellington NZ.

Dang, A.N., Jackson, B.M., Benavidez, R., Tomscha, S.A., 2021. Review of ecosystem service assessments: Pathways for policy integration in Southeast Asia. Ecosyst. Serv. https://doi.org/10.1016/j.ecoser.2021.101266.

Davidson, N.C., 2014. How much wetland has the world lost? Long-term and recent trends in global wetland area. Mar. Freshw. Res. 65, 934–941. https://doi.org/10.1071/MF14173.

Di, H.J., Cameron, K.C., 2002. Nitrate leaching in temperate agro-ecosystems: source, factors, and mitigation strategies. Nutr. Cycl. Agroecosys 64, 237–256.

Diaz, R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. Science 1979 (321), 926–929. https://doi.org/10.1126/science.1156401.

Díaz, S., Zafra-Calvo, N., Purvis, A., Verburg, P.H., Obura, D., Leadley, P., Chaplin-Kramer, R., de Meester, L., Dulloo, E., Martín-López, B., Shaw, M.R., Visconti, P., Broadgate, W., Bruford, M.W., Burgess, N.D., Cavender- Bares, J., DeClerck, F., Fernández-Palacios, J.M., Garibaldi, L.A., Hill, S.L.L., Isbell, F., Khoury, C.K., Krug, C.B., Liu, J., Maron, M., McGowan, P.J.K., Pereira, H.M., Reyes-García, V., Rocha, J., Rondinini, C., Shannon, L., Shin, Y.J., Snelgrove, P.V.R., Spehn, E.M., Strassburg, B., Subramanian, S.M., Tewksbury, J.J., Watson, J.E.M., Zanne, A.E., 2020. Set ambitious goals for biodiversity and sustainability. Science 1979 (370), 411–413. https://doi.org/10.1126/science.abe1530.

Dodds, W.K., Jones, J.R., Welch, E.B., 1998. Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Resour. Manag. 32, 1455–1462.

Ekroos, J., Ödman, A.M., Andersson, G.K.S., Birkhofer, K., Herbertsson, L., Klatt, B.K., Olsson, O., Olsson, P.A., Persson, A.S., Prentice, H.C., Rundlöf, M., Smith, H.G., 2016. Sparing land for biodiversity at multiple spatial scales. Front. Ecol. Evol. 3, 1–11. https://doi.org/10.3389/fevo.2015.00145.

Forestry, O. of the M. of, 2018. The One Billion Tree Program- Actions and Decisions for Implementation.

Garibaldi, L.A., Oddi, F.J., Miguez, F.E., Bartomeus, I., Orr, M.C., Jobbágy, E.G., Kremen, C., Schulte, L.A., Hughes, A.C., Bagnato, C., Abramson, G., Bridgewater, P., Carella, D.G., Díaz, S., Dicks, L.V., Ellis, E.C., Goldenberg, M., Huaylla, C.A., Kuperman, M., Locke, H., Mehrabi, Z., Santibañez, F., Zhu, C.D., 2021. Working landscapes need at least 20% native habitat. Conserv. Lett. 14, 1–10. https://doi.org/10.1111/conl.12773.

Gergel, S.E., 2005. Spatial and non-spatial factors: When do they affect landscape indicators of watershed loading? Landsc. Ecol. 20, 177–189. https://doi.org/10.1007/s10980-004-2263-y.

Gilby, B.L., Olds, A.D., Brown, C.J., Connolly, R.M., Henderson, C.J., Maxwell, P.S., Schlacher, T.A., 2021. Applying systematic conservation planning to improve the allocation of restoration actions at multiple spatial scales. Restor. Ecol. 29 https://doi.org/10.1111/rec.13403.

Hare, K.M., Borrelle, S.B., Buckley, H.L., Collier, K.J., Constantine, R., Perrott, J.K., Watts, C.H., Towns, D.R., 2019. Intractable: species in New Zealand that continue to decline despite conservation efforts. J. R. Soc. N. Z. 49, 301–319. https://doi.org/10.1080/03036758.2019.1599967.

Häyhä, T., Franzese, P.P., 2014. Ecosystem services assessment: a review under an ecological-economic and systems perspective. Ecol. Model. 289, 124–132. https://doi.org/10.1016/j.ecolmodel.2014.07.002.

Hillebrand, H., Donohue, I., Harpole, W.S., Hodapp, D., Kucera, M., Lewandowska, A.M., Merder, J., Montoya, J.M., Freund, J.A., 2020. Thresholds for ecological responses to global change do not emerge from empirical data. Nat. Ecol. Evol. 4, 1502–1509. https://doi.org/10.1038/s41559-020-1256-9.

IPBES, 2019. Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. 10.5281/ZENODO.6417333.

IPCC, 2022. Climate Change 2022: Mitigation of Climate Change.

Jackson, B., Pagella, T., Sinclair, F., Orellana, B., Henshaw, A., Reynolds, B., Mcintyre, N., Wheater, H., Eycott, A., 2013. Polyscape: a GIS mapping framework providing efficient and spatially explicit landscape-scale valuation of multiple ecosystem services. Landsc. Urban Plan. 112, 74–88. https://doi.org/10.1016/j.landurbplan.2012.12.014.

Jellinek, S., Parris, K.M., Driscoll, D.A., Dwyer, P.D., 2013. Are incentive programs working? Landowner attitudes to ecological restoration of agricultural landscapes. J. Environ. Manage. 127, 69–76. https://doi.org/10.1016/j.jenvman.2013.04.034.

Landcare Research, 2015. Fundamental Soil Layers New Zealand Soil Classification. Lincoln, New Zealand.

Landcare Research, 2020. New Zealand Landcover Database (LCDB5 v5).

Linz, 2013. Wellington LiDAR 1m DEM. Wellington, New Zealand.

Lobell, D.B., Schlenker, W., Costa-Roberts, J., 2011. Climate trends and global crop production since 1980. Science 1979 (333), 616–621. https://doi.org/10.3726/978-3-653-06341-7/62.

McDowell, R.W., Hedley, P.P., Rissmann, C., Catto, W., Patrick, W., 2019. Why are median phosphorus concentrations improving in New Zealand streams and rivers? J. R. Soc. N. Z. 49, 143–170. https://doi.org/10.1080/03036758.2019.1576213.

MfE, 2021. New Zealand's Greenhouse Gas Inventory Snapshot 1990–2019 [WWW Document]. URL https://environment.govt.nz/publications/new-zealands-greenhouse-gas-inventory-1990-2019-snapshot/how-new-zealand-compares-to-other-countries (accessed 10.13.22).

Ministry for the Environment, 2010. Legally Protected Conservation Land in New Zealand.

Ministry for the Environment, 2020. Guidance on look-up tables for setting nutrient targets for periphyton.

Ministry for the Environment, 2021. New Zealand's Greenhouse Gas Inventory 1990-2019. Wellington NZ.

Mitsch, W.J., Gossilink, J.G., 2000. The value of wetlands: Importance of scale and landscape setting. Ecol. Econ. 35, 25–33. https://doi.org/10.1016/S0921-8009(00)00165-8.

Myers, S.C., Clarkson, B.R., Reeves, P.N., Clarkson, B.D., 2013. Wetland management in New Zealand: are current approaches and policies sustaining wetland ecosystems in agricultural landscapes? Ecol. Eng. 56, 107–120. https://doi.org/10.1016/j.ecoleng.2012.12.097.

Nahlik, A.M., Fennessy, M.S., 2016. Carbon storage in US wetlands. Nat. Commun. 7, 1–9. https://doi.org/10.1038/ncomms13835.

Newbold, S.C., 2005. A combined hydrologic simulation and landscape design model to prioritize sites for wetland restoration. Environ. Model. Assess. 10, 251–263. https://doi.org/10.1007/s10666-005-9002-x.

NIWA, 2018. Virtual climate station data and products. Wellington NZ.

Paris Agreement to the United Nations Framework Convention on Climate Change (No. T.I.A.S. No. 16-1104), 2015.

Parliamentary Counsel Office, 2020. Resource Management (Stock Exclusion) Regulations 2020 12.

Paul, E.A., Clark, F.E., 1996. Soil Microbiology and Biochemistry, 2nd ed. Academic Press, London.

Qian, S.S., Cuffney, T.F., 2012. To threshold or not to threshold? That's the question. Ecol. Ind. 15, 1–9. https://doi.org/10.1016/j.ecolind.2011.08.019.

Rajanayaka, C., Weir, J., Barkle, G., Griffiths, G., Hadfield, J., 2020. Assessing changes in nitrogen contamination in groundwater using water aging: Waikato River, New Zealand. J. Contam. Hydrol. 234, 103686 https://doi.org/10.1016/j.jconhyd.2020.103686.

Ramankutty, N., Mehrabi, Z., Waha, K., Jarvis, L., Kremen, C., Herrero, M., Rieseberg, L. H., 2018. Trends in global agricultural land use: implications for environmental health and food security. Annu. Rev. Plant Biol. 69, 789–815. https://doi.org/10.1146/annurev-arplant-042817-040256.

Richards, J., Chambers, T., Hales, S., Joy, M., Radu, T., Woodward, A., Humphrey, A., Randal, E., Baker, M.G., 2021. Nitrate contamination in drinking water and colorectal cancer: exposure assessment and estimated health burden in New Zealand. Environ. Res. 204, 112322 https://doi.org/10.1016/j.envres.2021.112322.

RMA, 2020. Resource Management (Stock Exclusion) Regulations 2020. New Zealand.

Ruamahanga Whaitua Committee, 2018. Ruamāhanga Whaitua Implementation Programme.

Scholes, R.J., Reyers, B., Biggs, R., Spierenburg, M.J., Duriappah, A., 2013. Multi-scale and cross-scale assessments of social-ecological systems and their ecosystem services. Curr. Opin. Environ. Sustain. 5, 16–25. https://doi.org/10.1016/j.cosust.2013.01.004.

Sharples, K., Firth, M., Hinder, V., Hill, A., Jeffery, M., Sarfati, D., Brown, C., Atmore, C., Lawrenson, R., Reid, P., Derrett, S., Macapagel, J., Keating, J., Secker, A., De Groot, C., Christopher Jackson, M.F., 2018. The New Zealand PIPER Project: colorectal cancer survival according to rurality, ethnicity and socioeconomic deprivation- results from a retrospective cohort study. N. Z. Med. J. 131, 1175–9716.

Singh, Gourevitch, J.D., Wemple, B.C., Watson, K.B., Rizzo, D.M., Polasky, S., Ricketts, T. H., 2019. Optimizing wetland restoration to improve water quality at a regional scale. Environ. Res. Lett. 14 https://doi.org/10.1088/1748-9326/ab1827.

Smith, P., Davies, C.A., Ogle, S., Zanchi, G., Bellarby, J., Bird, N., Boddey, R.M., McNamara, N.P., Powlson, D., Cowie, A., van Noordwijk, M., Davis, S.C., Richter, D. D.B., Kryzanowski, L., van Wijk, M.T., Stuart, J., Kirton, A., Eggar, D., Newton-Cross, G., Adhya, T.K., Braimoh, A.K., 2012. Towards an integrated global framework to assess the impacts of land use and management change on soil carbon: current capability and future vision. Glob. Chang. Biol. 18, 2089–2101. https://doi.org/10.1111/j.1365-2486.2012.02689.x.

Snelder, T.H., Moore, C., Kilroy, C., 2019. Nutrient concentration targets to achieve periphyton biomass objectives incorporating uncertainties. J. Am. Water Resour. Assoc. 55, 1443–1463. https://doi.org/10.1111/1752-1688.12794.

Spake, R., Barajas-barbosa, M.P., Blowes, S.A., Bowler, D.E., Callaghan, C.T., Garbowski, M., Jurburg, S.D., van Klink, R., Korell, L., Ladouceur, E., Rozzi, R., Viana, D.S., Xu, W., Chase, J.M., 2022. Detecting thresholds of ecological change in the anthropocene. Ann. Rev. Environ. Resources In Press 1–25.

Stats NZ, 2018. 2018 Census data Retrieved from www.stats.govt.nz.

Stats NZ, 2021. More land on Māori farms used for forest plantation [WWW Document]. URL https://www.stats.govt.nz/news/more-land-on-maori-farms-used-for-forest-plantation/ (acccessed 9.8.22).

Strassburg, B.B.N., Beyer, H.L., Crouzeilles, R., Iribarrem, A., Barros, F., de Siqueira, M. F., Sánchez-Tapia, A., Balmford, A., Sansevero, J.B.B., Brancalion, P.H.S., Broadbent, E.N., Chazdon, R.L., Filho, A.O., Gardner, T.A., Gordon, A., Latawiec, A., Loyola, R., Metzger, J.P., Mills, M., Possingham, H.P., Rodrigues, R.R., Scaramuzza, C.A.de.M., Scarano, F.R., Tambosi, L., Uriarte, M., 2019. Strategic approaches to restoring ecosystems can triple conservation gains and halve costs. Nat. Ecol. Evol. 3, 62–70. https://doi.org/10.1038/s41559-018-0743-8.

Taura, Y., Van-Schravendijk- Goodman, C., Clarkson, B.R., 2017. Te Reo o Te Repo.

Taylor, A.I., 2018. Evaluating the sensitivity of the LUCI model to GIS datasets to enable robust farm management decisions in New Zealand.

The LUCI team, 2019. LUCI Help Documentation.

Theodore, T.H., 2010. Micronutrient deficiency conditions: global health issues. Public Health Rev. 32, 243–255.

Thomas, A., Masante, D., Jackson, B., Cosby, B., Emmett, B., Jones, L., 2020. Fragmentation and thresholds in hydrological flow-based ecosystem services. Ecol. Appl. 30, 1–14. https://doi.org/10.1002/eap.2046.

Tomscha, S.A., Bentley, S., Platzer, E., Jackson, B., de Roiste, M., Hartley, S., Norton, K., Deslippe, J.R., 2021. Multiple methods confirm wetland restoration improves ecosystem services. Ecosyst. People 17, 25–40. https://doi.org/10.1080/26395916.2020.1863266.

Tomscha, S., Deslippe, J., de Róiste, M., Hartley, S., Jackson, B., 2019. Uncovering the ecosystem service legacies of wetland loss using high-resolution models. Ecosphere 10. https://doi.org/10.1002/ecs2.2888.

Tran, D.X., Pearson, D., Palmer, A., Lowry, J., Gray, D., Dominati, E.J., 2022. Quantifying spatial non-stationarity in the relationship between landscape structure and the provision of ecosystem services: an example in the New Zealand hill country. Sci. Total Environ. 808, 152126 https://doi.org/10.1016/j.scitotenv.2021.152126.

Trodahl, M., Burkitt, L., Bretherton, M., Deslippe, J., Jackson, B., Metherell, A., 2017. Developing N & P Export Coefficients for Rural Landscape Modelling in LUCI. Fertilizer and Lime Research Centre (FLRC) Workshop Proceedings Occasional Report 9.

Trodahl, M.I., Jackson, B.M., Deslippe, J.R., Metherell, A.K., 2017b. Investigating trade-offs between water quality and agricultural productivity using the Land Utilisation and Capability Indicator (LUCI)–A New Zealand application. Ecosyst. Serv. 26, 388–399. https://doi.org/10.1016/j.ecoser.2016.10.013.

UNEP, 2020. Indicators for the post-2020 global biodiversity framework.

UNEP-WCMC, 2021. Protected areas map of the world [WWW Document]. URL www.protectedplanet.net.

Wang, N., Mitsch, W.J., 1998. Estimating phosphorus retention of existing and restored coastal wetlands in a tributary watershed of the Laurentian Great Lakes in Michigan, USA. Wetl. Ecol. Manag. 6, 69–82. https://doi.org/10.1023/A:1008451823394.

Ward, M., Saura, S., Williams, B., Ramírez-Delgado, J.P., Arafeh-Dalmau, N., Allan, J.R., Venter, O., Dubois, G., Watson, J.E.M., 2020. Just ten percent of the global terrestrial protected area network is structurally connected via intact land. Nat. Commun. 11, 1–10. https://doi.org/10.1038/s41467-020-18457-x.

Waring, S., 2017. The factors that influence the reestablishement of *Potocarpus totara* (totara) and *Dacrycarpus dacrydioides* (kahikatea) in a freshwater New Zealand wetland. Victoria University of Wellington.

White, M., Harmel, D., Yen, H., Arnold, J., Gambone, M., Haney, R., 2015. Development of sediment and nutrient export coefficients for U.S. Ecoregions. J. Am. Water Resour. Assoc. 51, 758–775. https://doi.org/10.1111/jawr.12270.

Woodley, S., Locke, H., Laffoley, D., Mackinnon, K., Sandwith, T., Smart, J., 2019. A review of evidence for area-based conservation targets for the post-2020 global biodiversity framework. Parks 25, 31–46. https://doi.org/10.2305/IUCN.CH.2019.PARKAS-25-2SW2.en.

Zedler, J.B., Kercher, S., 2005. WETLAND RESOURCES: status, trends, ecosystem services, and restorability. Annu. Rev. Env. Resour. 30, 39–74. https://doi.org/10.1146/annurev.energy.30.050504.144248.