Manaaki Whenua
Landcare Research

Carbon sequestration potential of non-ETS land on farms

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Carbon sequestration potential of non-ETS land on farms

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Summary

Project and client

- To help build a robust evidence base on which the agricultural sector can identify potential on-farm emissions reductions, MPI contracted Manaaki Whenua – Landcare Research (Manaaki Whenua) in collaboration with SCION, NIWA and AgResearch to review the carbon sequestration potential of currently non-ETS vegetation classes on farmland. These include wetlands, riparian strips, pole plantings, shelterbelts, and retired land that is not eligible for ETS[1].

Objectives

- Review the carbon sequestration potential of a set of on-farm activities that do not meet the forest land criteria of the current ETS.
- Describe the evidence base and background relating to management of these activities that would influence their inclusion as viable on-farm carbon sinks
- Provide a summary of known literature and New Zealand data on emissions factors associated with those activities.
- Provide example farm case studies where possible and comment on drivers and barriers to these potential sequestration activities.

Methods

- Manaaki Whenua, SCION, NIWA and AgResearch (the authors) carried out a desk-based exercise to summarise the characteristics, background and management of each specific vegetation class.
- Preliminary findings were discussed at a workshop held between the authors, the Biological Emissions Reference Group (BERG) and Government agencies (MPI, MfE).
- We searched the literature to review sequestration potential and emissions factors for each class and considered specific New Zealand data and literature to assess sequestration potential for the various target on-farm activities.
- Finally, we considered example case studies where possible and commented on drivers and barriers on farms in relation to the practical application of these potential sequestration activities.

Results

- There is a lack of New Zealand sequestration estimates and emission factors for all the specific vegetation classes contemplated in this report. Reported values given are surrogates based on different vegetation types, and the associated uncertainty is likely substantial.

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[1] The Terms of Reference for this study focus on biomass emissions only and exclude non-CO_2 GHGs, soil C and associated processes, other than for Wetlands.

- Approximate potential sequestration rates were highest for small lots and shelterbelts (6.5–26.3 $t{\cdot}CO_2e{\cdot}ha^{-1}yr^{-1}$), intermediate for riparian strips (0–5.28 $t{\cdot}CO_2e{\cdot}ha^{-1}yr^{-1}$) and pole planting (2.0 $t{\cdot}CO_2e{\cdot}ha^{-1}yr^{-1}$) and lowest for natural wetlands (0–2.0 $t{\cdot}CO_2e{\cdot}ha^{-1}yr^{-1}$) and non-ETS compliant retired land (0–0.47 $t{\cdot}CO_2e{\cdot}ha^{-1}yr^{-1}$).
- Potential new C sinks have finite longevity and the approximate period of sequestration of the different vegetation types is suggested to be 20-30 years, except for natural wetlands and retired land where sequestration may be low but could potentially extend for longer (c. 50 years or more).
- Small Woodlots nationally could sequester c. 1 $Mt{\cdot}CO_2e{\cdot}yr^{-1}$ sequestration, until they mature about 2050, if established on a little over 4 ha of every 1000 ha. (0.4 % of agricultural land).
- For shelterbelts, the scope for net increases is expected to be limited and will face ETS implementation challenges. In the case of new small lots, riparian belts and pole planting significant new sinks could be created if sufficient areas can be permanently set aside. For example, basic calculations imply that a national C sink created by a 10m wide riparian strip on 50% of New Zealand streams and rivers adjoining farmland could amount to 0.72$Mt{\cdot}CO_2e{\cdot}yr^{-1}$ although this rate should be treated with caution as it is based on several unproven assumptions.
- Wetlands could sequester c. 0.06 $Mt{\cdot}CO_2e{\cdot}yr^{-1}$ nationwide on farms if agricultural land in peat mires was retained. Wetland net CO_2e exchange is the balance of C sequestration and CH_4 and N_2O emissions that are complex to measure. Sequestration by constructed wetlands is unknown but may be close to zero due to CH_4 and N_2O emissions.
- Farm Case Study results for three existing livestock farming operations indicate that sequestration by current and planned vegetation could account for c. 0 – 2.5% of current emissions by livestock on intensive sheep and dairy farms, and c. 5 - 20% on the hill sheep/beef example.

Conclusions

- Most of the vegetation classes considered here, (riparian strips, pole planting, shelterbelts, and small woodlots) are widespread and common on farmland in some regions of New Zealand, are already accepted practices, and as such could contribute directly to GHG sequestration benefits to New Zealand farms. In practise the intensive sheep and dairy farm case-study examples investigated indicate that offsets from these vegetation classes found in these two cases may compensate for only a minor proportion (0 – 2.5%) of overall emissions from these farms under current practises. For the sheep and beef farm case-study investigated offsets from these vegetation classes accounted for a significant proportion (c. 5-20%) of overall emissions under current practice.
- In some cases (woodlots, shelterbelts, riparian) and on some landscapes (e.g. hill and steepland) there is potential for increased sequestration, but that is limited unless it is associated with a major targeted initiative, significantly increased land area and a favourable cost/benefit ratio.
- Realizing any potential will require quantifying spatial extent, sequestration rates, longevity of the sink, defining a time baseline, and adoption into a New Zealand

scheme, while addressing actual and perceived barriers by farmers. This will not be simple. Only a deliberate, additional national initiative aimed at increasing the land area of those classes will realise any sizable C sequestration.

- Biomass carbon stocks (e.g. standing forest, peat mires, soil carbon) of themselves do not create any offset benefit. Only net change (increase) in those stocks becomes an offset sink (sequestration).
- Biomass carbon sinks are temporary (they saturate once vegetation growth and mortality/decomposition are balanced is a steady-state condition, while the many other co-benefits (water purification, shelter, erosion control, etc) are on-going. In most cases assessed here, the longevity of the sink will be <30 years.
- Actual sequestration rates, longevity of the sink and baselines specific for all these classes are unknown but are critical to their use as an offset sink. There is a deficiency of data and statistical estimates of carbon stocks and sequestration potential for all non-ETS compliant on-farm units considered in this report. Surrogates of carbon stocks and sequestration potential used here have been taken from related contexts but are expected to be only partly comparable and not fully adequate (e.g. riparian strips host different vegetation structures and growth conditions compared with typical successional indigenous forest).
- There are no adequate estimates of spatial extent and change through time of these vegetation classes on New Zealand farmland as a whole, and as yet no standardised means of quantifying them.
- The multiple conditions and complexities associated with each of these units (e.g. major differences between mires/peatlands, marshes and constructed wetlands, or, between shelterbelts of different species and whether or not they are hedged/mown) imply a large level of uncertainty and wide range of variability in their C stocks and sequestration potential. This variability and uncertainty could be reduced with the specification of particular conditions (e.g. peatlands instead of wetlands).
- Knowledge gaps can be addressed by future research and we recommend for any potentially viable classes that the first step is to quantify actual emissions factors for these offsets by undertaking specific design studies. We note that related questions have been addressed for some time and will be (i) methodologically challenging (e.g. tracing the flow of various biomass compartments below-ground or in water), (ii) may require time to be answered given the pace of some natural processes or use of a space-for-time approach, and (iii) may pose challenges for implementation and monitoring.
- The on-farm classes considered in this report contribute considerable associated co-beneficial farm, ecosystem, and biodiversity services beyond carbon sequestration. They are already identified - at least to some extent - in farm environmental plans and it would be technically feasible for a motivated farmer to map and classify the vegetation on their own land.
- Next steps towards development of comprehensive emissions factors for each class and developing systems for mensuration, assessment and monitoring on individual farms is needed by commissioning specific design studies with three main objectives for each vegetation class. i) Estimate the extent of the resource on farmland across New Zealand, ii) Develop comprehensive emissions factors that account for variation within the vegetation class across the country and through time, and iii) Develop

mensuration techniques that can be used to assess the biomass and sequestration of each class on farmland. Objective ii) would form the basis for development of appropriate lookup tables.

Recommendations

- Before any farm can consider increasing their biomass offset sinks they will need to know the related cost/benefit ratio. What is the cost of production of C-units compared with livestock units? We recommend that an economic analysis is carried out and that it considers the relationship between the areas of farmland that would be needed to offset typical farm emissions.
- It is recommended that the results presented here for the on-farm vegetation classes are considered in relation to the likelihood of any of these classes being adopted as an expanded, additional, significant new C offset sink on farms beyond business-as-usual land-use.
- Most of the vegetation classes addressed in this report have potential to sequester significant C, yet there is a deficiency of data and statistical estimates of actual carbon stocks, potential sequestration (and variability) and longevity of the sink for all non-ETS compliant on-farm units considered in this report. For these knowledge gaps to be filled for the sake of both New Zealand farms, and national accounts, will require several specific quantitative investigations of actual extent and sequestration to be commissioned that may be associated with existing research programmes.

Glossary

ETS – New Zealand Emissions Trading Scheme

UNFCCC – United Nations Framework Convention on Climate Change

LULUCF – Total emissions and removals from activities relating to Land use, Land-use change and Forestry (from the following categories: forest land, cropland, grassland, wetlands, settlements and other land).

IPCC – Intergovernmental Panel on Climate Change.

GHG – Greenhouse Gases (carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), fluorinated gases)[2]

CO_2e – carbon dioxide equivalent emissions is a common scale for comparing emissions of different GHGs[3]

EF – An emission factor is defined as the average emission rate of a given GHG for a given source, relative to units of activity.

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[1]IPCC, 2014: Annex II: Glossary [Mach, K.J., S. Planton and C. von Stechow (eds.)]. In: Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, pp. 117-130.

[3]https://unfccc.int/process/transparency-and-reporting/greenhouse-gas-data/greenhouse-gas-data-unfccc/definitions

1 Introduction

The Biological Emissions Reference Group (BERG) of government and industry members representing farming land-owners has asked whether any net vegetation biomass carbon sink is created by widespread on-farm activities involving vegetation establishment and growth but which do not meet the requirements of the existing ETS. The classes of vegetation specific to this review are: wetlands, riparian strips, pole planting, shelterbelts, small woodlots, and revegetation of retired land not otherwise eligible for ETS[4]. Farmers looking to offset their total farm emissions need to understand all potential greenhouse gases (GHG) sources and sinks on their properties.

Carbon sequestration in non-forest ecosystems has been outside New Zealand's Emissions Trading Scheme (ETS) as these ecosystems did not meet conditions for compliance markets for carbon (C) originally adopted by New Zealand under the Kyoto Protocol for the first commitment period (CP1: 2008–2012). During this period New Zealand accounted internationally for all Article 3.3 activities (i.e. new forests that had been established since 1989 and all deforestation). Countries could voluntarily account for other activities under Article 3.4 (management effects), which included accounting for net emissions from vegetation and soil from forest management (of forests that existed prior to 1990), grazing land management and cropland management. New Zealand chose not to account for these voluntary activities during CP1. For the second commitment period (CP2: 2013-2020), accounting for forest management under Article 3.4 became mandatory and an additional voluntary activity was included for wetland management. While New Zealand's emissions reduction target for the period is now taken under the UNFCCC, it applies the Kyoto Protocol framework of rules to accounting, and continues to account only for the mandatory forestry activities.

New Zealand subsequently signed up to the Paris Agreement in December 2015 (ratified October 2016), which allows countries to develop their own nationally determined contribution (NDC) to reduce national emissions and adapt to the impacts of climate change. The Paris Agreement requires each Party to prepare, communicate and maintain successive NDCs that it intends to achieve with the aim of increasing ambition over time. To achieve those goals New Zealand is exploring and considering whether there are significant unaccounted carbon emissions in non-forest land uses.

C emission rates for all land uses (emissions factors – EFs) represent the net above-ground and below-ground greenhouse gas exchange (expressed as carbon or carbon-dioxide equivalents) from vegetation and soil per unit time as a result of biogenic processes. IPCC (2006) Guidelines[5] provide some general EFs in many cases, often at a national level, but suggest that best practice is to develop unbiased specific in-country EFs.

[4] This review only considers vegetation and excludes soil C, except for Wetlands.

[5] IPCC 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. (https://www.ipcc-nggip.iges.or.jp/public/2006gl/)

2 Background

New Zealand farmers are investigating the GHG emissions associated with their operations. Large stocks of C exist in vegetation and in soils on farmland, but it is the net GHG exchange between the land and the atmosphere that is key to understanding whether they are a net sink or source. Thus, the task involves knowing and accounting for all potential C sources and sinks on a property. Findings are presented as emissions factors (EFs), and are normally presented as the summed equivalent of all GHGs in $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$.

Carbon offset sinks and any related EFs need to meet verifiable standards. According to IPCC (2006) Guidelines that means they need to comply with agreed *good practice*, preferably be country-specific, and be unbiased. To devolve national EFs to an individual property level requires verifiable standards for C. Those are often spelled out by C markets as needing to be real, measurable, permanent, additional, and independently verifiable (e.g. VCS-Verified Carbon Standard (http://verra.org/), ICROA-International Carbon Reduction & Offset Alliance (http://www.icroa.org/Quality-Assurance)).

Net sequestration occurs as vegetation develops from a low biomass condition to a higher biomass condition per unit area over time, and is finite. The C accumulation curve from an 'empty' to a 'full' or mature condition is normally sigmoidal in shape, and actual sequestration rates vary with time along that curve. Rates tend to be slow initially (when plants are small and space is not fully occupied), then accelerate to a peak rate, and eventually decline again (once maximum biomass stocks are approached) to a relatively stable state when new growth is matched by mortality and there is virtually no net sequestration. The length of periods of net sequestration (longevity of the sink) will depend on the potential biomass that can be accumulated on a site (i.e. the asymptote of the C accumulation curve), and the size and shape of those curves is different for each vegetation class. The rate of sequestration, size of the mature C stocks, and time to maturity achieved by established herbaceous vegetation will be relatively quick and small compared with those processes for forests, which can develop huge C stocks and continue to sequester C for centuries. Carbon accumulation processes are naturally variable and determined by multiple biotic and abiotic factors so cannot normally be generalised from a single or even a few cases. Soil is a very important C pool currently excluded from the ETS, but needs to be considered in relation to long-term land-use change and new biomass offset sinks. Soil C has been excluded from this investigation, with the exception of wetlands where peat is treated as soil.

This report uses standard units of $\mathbf{t.CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}}$ for GHG-equivalent carbon sequestration wherever possible. In a few places **Mt** (mega-tonnes) $\mathbf{CO_2e{\cdot}yr^{-1}}$ is used for national or global estimates.

3 Objectives

- Review the carbon sequestration potential of a set of on-farm activities that do not meet the forest land criteria of the current ETS.
- Describe the evidence base and background relating to management of these activities that would influence their inclusion as viable on-farm carbon sinks
- Provide a summary of known literature and New Zealand data on emissions factors associated with those activities.
- Provide examples of case studies where possible and comment on drivers and barriers to these potential sequestration activities.

4 Methods

The scientific expertise of Manaaki Whenua-LR, SCION, NIWA and AgResearch pertinent to the different subject vegetation classes was combined in a desk-based exercise, and a workshop with BERG representatives, to summarise the characteristics, background and management in New Zealand of each specific vegetation class. We searched for additional literature to review the sequestration potential and EFs for each class. We then considered the New Zealand data and literature to assess on-farm sequestration potential for the various target on-farm activities relating to wetlands, riparian strips, pole planting, shelterbelts, small lots, and revegetation of retired land not otherwise eligible for ETS. The order in which these activities are presented in the report simply reflects the schedule of contracted services and has no relationship to their relative potential as carbon offset sinks. In practise there is overlap between many of the activities – e.g. wetlands and riparian planting, woodlots and shelter belts, woodlots and wide-spaced plantings, etc. Finally, we considered example case studies where possible and commented on drivers and barriers on farms in context of the practical application of these potential sequestration activities.

5 Results

A. Wetlands (John Quinn & Larry Burrows)

a) Definition, characteristics, background and management

Wetlands are defined as permanently or intermittently wet areas, shallow water, or land/water margins that support a natural community of plants and animals adapted to living in wet conditions. They occur as ephemeral wetland, pakihi, swamp, fen, bog, or marsh types (Johnson & Gerbeaux 2004), several of which are successional forms (Figure A1), but have quite different carbon emission profiles.

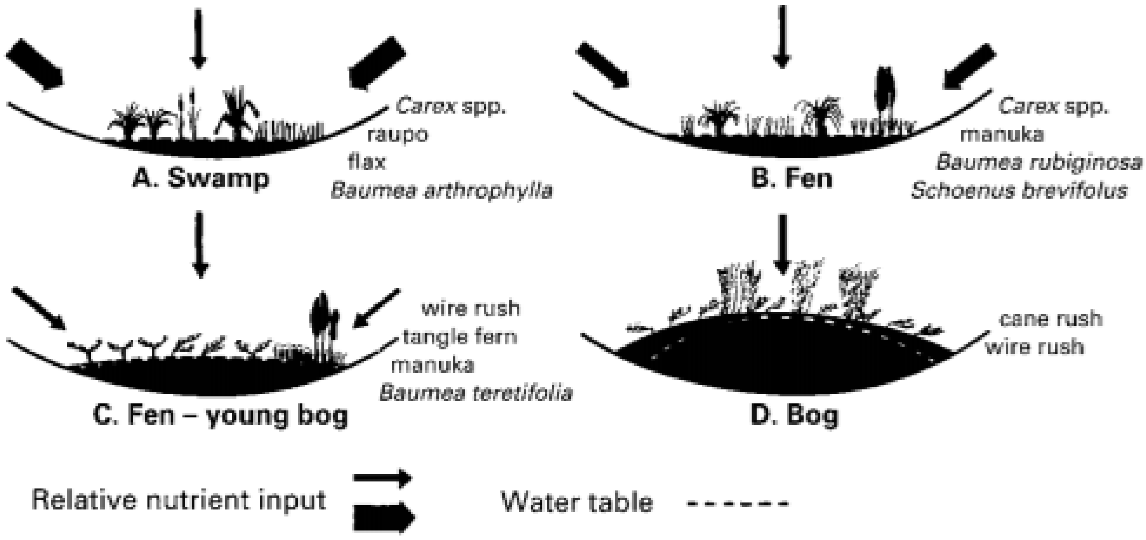

Figure A 1: The evolution of Waikato wetlands over 10,000 years (from Johnson & Gerbeaux 2004).

Wetlands are important sinks of carbon (C) due to their high biomass productivity combined with low organic matter decomposition rates, resulting from anaerobic conditions in waterlogged soils. Emissions of the greenhouse gases methane (CH_4) and nitrous oxide (N_2O) produced under the same anaerobic conditions reduce their net effect as a C sink. Wetlands occupy 5–8% of the Earth's land surface (Mitsch & Gosselink 2007) but contain a disproportionately large (20–30%) proportion of global soil C ($1.55x10^6$ Mt) (Lal 2008).

Freshwater wetlands are classified more broadly as either mires (peatlands with organic soils) or marshes (freshwater mineral soil wetlands), based on their organic C content. Mires accumulate C as peat because organic matter production and accumulation are greater than decomposition under anaerobic conditions. In marshes, C is sequestered by both sediment deposition from upstream catchments and *in situ* biomass production that may exceed C decomposition rates.

Mires (peatlands) store more C per unit area than any other terrestrial biome (Dise & Phoenix 2011). Despite only covering 2–3% of the global land area, they are estimated to store 6.12×10^6 Mt C (Yu 2011), equivalent to more than half the carbon dioxide (CO_2) currently stored in the atmosphere (Dise 2009). Chapman et al. (2012) have estimated that restoration of peatlands could provide up to 2.7 Mt CO_2e (carbon dioxide equivalent of all GHGs) savings per year in Scotland (cited in Ratcliffe et al. 2017).

It is important to note, however, that disturbance factors such as fire or drought can cause peatlands to abruptly lose C that had previously been retained for hundreds of years (Frolking et al. 2014). Such losses may account for the observation that contemporary C sequestration rates measured by gas exchange methodologies are often several times higher (9- to 22- fold) than long-term rates based on C cores and dating (Campbell et al. 2014; Ratcliffe et al. 2017).

Wetlands provide a wide range of valuable ecosystem services apart from C sequestration (Clarkson et al. 2013), some of which are summarised in Figure A2. Wetlands on farms are often referred to as 'catchment kidneys' or 'nature's kidneys' due to their ability to clean water. They do this by retaining and transforming contaminant inputs by settling and storing particulates (and associated nutrients, agrichemicals and pathogens), removing pathogens by solar radiation exposure in open water and grazing by zooplankton, and reducing nitrogen export in water by denitrification of nitrate to nitrogen gases and uptaking nutrients into plant biomass. Wetlands also influence the hydrological cycle through water storage and evapotranspiration, with effects varying between wetland types (Bullock & Acreman 2003).

Location-specific factors are important, but upland wetlands generally tend to be flood-generating areas while floodplain wetlands have a greater potential to moderate floods (Acreman & Holden 2013). Wetlands provide wildlife habitat, including taonga species for Māori (tuna/eels, adult inanga/whitebait species, and ducks) and enhance catchment-scale biodiversity (e.g. of zooplankton; Eivers et al. 2017). They also provide cultural resources for traditional Māori art and crafts (e.g. flax and raupō).

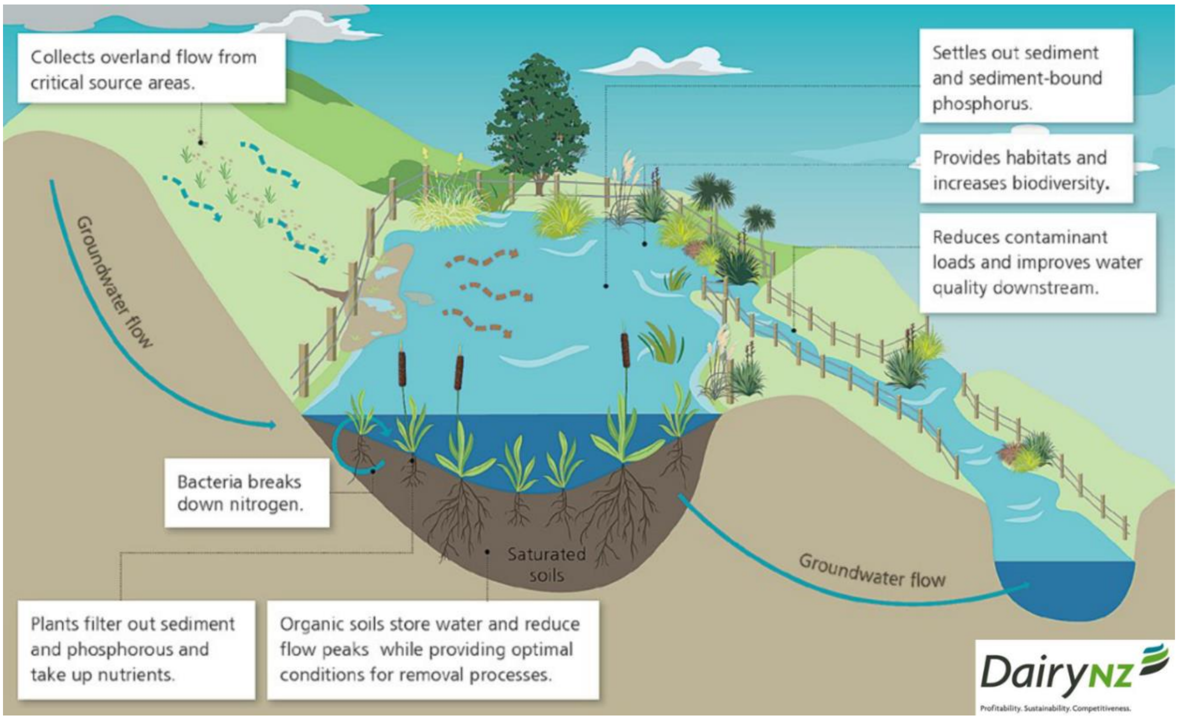

Figure A 2: A summary of the 'catchment kidney' ecosystem services of wetlands on farms for reducing contaminant inputs to downstream freshwater ecosystems. (Source: DairyNZ, www.dairynz.co.nz).

However, the predominant location of wetlands in low-lying flat areas of New Zealand (and elsewhere) has made them targets for draining, primarily for conversion to agricultural land. Since the arrival of European settlers the original extent of wetlands has been reduced by 90%, to approximately 250,000 ha (Ausseil et al. 2015). This loss continues, with 1,247 ha (0.5%) of total wetland area (comprising 214 individual wetlands) completely lost and another 5.4% (746 individual wetlands) experiencing partial loss over the period 2001–2016 (MfE & StatsNZ 2018). This wetland conversion has disturbed large stocks of C, and it is estimated that the current loss from 146,000 ha of farmed organic soils is between 0.5 and 2 $Mt{\cdot}CO_2{\cdot}yr^{-1}$, equivalent to 1–6% of the total greenhouse gas emissions from the New Zealand agriculture sector (Ausseil et al. 2015).

The Land Cover Data Base identifies over 7,000 wetlands (Class – Herbaceous Freshwater Vegetation), with 50% of the total area contained in 77 large wetlands, each over 500 ha, whereas 74% of wetlands are <10 ha. Most larger wetlands are on public land (MfE & StatsNZ 2018), but the vast majority of smaller wetlands are on private land surrounded by agricultural landscapes (Myers et al. 2013). Nevertheless, 7,500 ha of wetlands on private land are protected under QEII Trust covenants (MfE & StatsNZ 2018).

b) Carbon stocks in New Zealand wetlands

Ausseil et al. (2015) conducted the first national estimate of C stocks in freshwater wetlands based on the compilation of soil C data from 126 sites across New Zealand. Their results (Table A1) show that organic wetland types (mires) store c. 6-fold more C than mineral wetlands (marshes). Average peat depth in New Zealand mires is 3.9 m (Ausseil et al. 2015). In total, wetlands store 167 Mt·C (612 $Mt{\cdot}CO_2e$), which is equivalent to c. 10 times the national CO_2e emissions in 2010 (Ausseil et al. 2013). This indicates that wetlands are a significant store of what Nahlik and Fennessy (2016) term 'teal carbon' (cf. 'green carbon' stored in terrestrial biomass, and 'blue carbon' stored in the coast and ocean). This national C stock is substantially (28-fold) higher than the earlier estimate of Tate et al. (1997) of 22 $Mt{\cdot}CO_2e$, based on the New Zealand Land Resource Inventory, the National Soils Database, and conservative depth assumptions.

Table A 1: New Zealand wetland carbon storage estimates

Soil type	Class	Carbon content (grams of C per 100 g soil)	C stock (Mt)
Organic (mires)	Bog	49	102
	Fen	32	19
	Swamp	28	20
	Marsh	19	0.6
	Weighted average	34	144
Mineral (marshes)	Fen	10	3.5
	Swamp	6	8.3
	Marsh	9	5.8
	Pakihi	11	4.6
	Ephemeral	0.2	
	Weighted average	6	23

Adapted from Ausseil et al. 2015.

c) Annual wetland carbon sequestration

The rate of C sequestration in a wetland is the change in CO_2e storage, including emissions of the greenhouse gases methane (CH_4) and nitrous oxide (N_2O) as CO_2e (Figure A3). Wetlands also lose dissolved organic carbon to downstream aquatic ecosystems (Findlay et al. 2001), but the contribution of this to GHG will depend on how much gets converted to CH_4. Most of this dissolved organic C will be incorporated into aquatic food webs (Newbold et al. 1982; Rounick et al. 1982; Cole & Caraco 2001) and not lost to the atmosphere as CO_2. Arguably, C storage and CH_4 emission are the most important of these processes, because if N_2O emissions, due to incomplete denitrification of nitrate (to N_2), did not occur in the wetland, it may well occur in downstream aquatic ecosystems, after adding to eutrophication of streams, lakes and the ocean along the nitrogen cascade (Galloway et al. 2003).

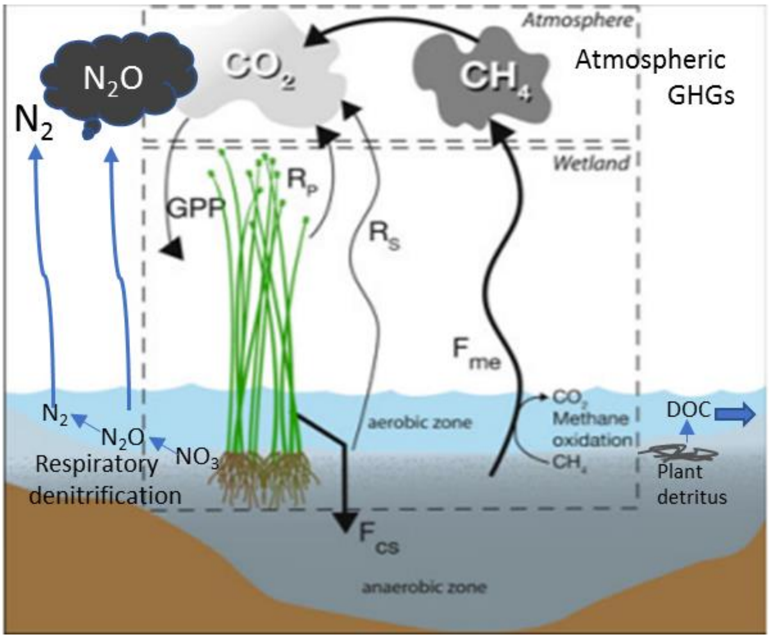

Figure A 3: Simplified conceptual model of carbon and nitrogen flows in a wetland and exchanges with the atmosphere.

Notes: F_{cs} = carbon sequestration; F_{me} = methane emissions; GPP = gross primary productivity; R_P = plant respiration; R_S = soil respiration; DOC = dissolved organic carbon.

Adapted from Mitsch et al. 2012.The gas clouds indicate the relative strength of the three greenhouses gases CO_2, CH_4 and N_2O. Conversion factors were 3.7 for C to CO_2 and 21 and 298 for CH_4 and N_2O, respectively (IPCC 2007). N_2 has no GHG effect.

Carbon sequestration in natural wetlands

The most comprehensive study of natural wetland C exchange in New Zealand (Goodrich et al. 2017) at the Kopuatai peat dome involved measuring CO_2 and CH_4 emissions (by eddy covariance measurements to estimate carbon balance at paddock scale) and estimating dissolved organic carbon export from a water balance over 4 years of dry, wet and normal summers (2012–2015). Kopuatai, located on the Hauraki Plains, in Waikato, is the largest remaining raised peat bog (mire) in New Zealand. Goodrich et al. (2017) found that the net ecosystem C balance was 135–217 $g \cdot C \cdot m^{-2} \cdot yr^{-1}$ (least for the year with a dry summer). This range is greater than rates for northern hemisphere bogs (−14 to 101 $g \cdot C \cdot m^{-2} \cdot yr^{-1}$), which they attributed to the mild Waikato climate and a shorter season of C loss, resulting in a longer growing season than in the northern hemisphere bog studies. Goodrich et al. (2017) calculated the Kopuatai annual global warming potential gas exchange (IPCC 1990) for the 4 years as −78.5, −43.2, −65.5, and −355.7 $g \cdot CO_2e \cdot m^{-2}$, equivalent to sequestration of 0.78–3.55 $t \cdot CO_2e \cdot ha^{-1} \cdot yr^{-1}$.

An earlier study (Campbell et al. 2014) used eddy covariance methods to measure CO_2 gas exchange only from Moanatuatua peat bog, located between Hamilton and Te Awamutu. This measured net ecosystem CO_2 exchanges of –218 and –250 $g \cdot C \cdot m^{-2} \cdot yr^{-1}$ (2.18–2.5 $t \cdot CO_2e \cdot ha^{-1} \cdot yr^{-1}$). Assuming low CH_4 emissions and typical dissolved organic carbon losses, Campbell et al. (2014) estimated the annual net ecosystem CO_2 balance at Moanatuatua was likely to have exceeded –200 $g \cdot C \cdot m^{-2} \cdot yr^{-1}$ for each of the 2 years, similar to the upper range from Kopuatai.

These contemporary rates were 6-fold higher than total C accumulation in a Moanatuatua bog measured since the Taupō volcanic eruption (about 1,900 years ago), averaging 34 $g \cdot C \cdot m^{-2} \cdot yr^{-1}$ (0.34 $t \cdot CO_2e \cdot ha^{-1} \cdot yr^{-1}$) (Schipper & McLeod 2002). Campbell et al. (2014) suggested the difference was likely to be due to intermittent large losses of C during fires, as well as the demonstrated successional changes to vegetation caused by fire affecting water balance.

Cooke et al. (2009) reviewed the significance of wetlands and riparian wetlands in the agricultural landscape as sources of N_2O emissions. The review used international studies due to the absence of New Zealand research. In wetlands receiving run-off from agricultural fields, rates are commonly in the range 0–45 $kg \cdot N \cdot ha^{-1} \cdot yr^{-1}$, with the high end of the range associated with high nitrate input and removal rates. Zero or low emission rates are associated with permanent inundation and low nitrate inputs.

Following are the key points from this review.

- There is a large range in the reported annual N_2O emission rates from riparian wetlands, from 0 to over 100 $kg \cdot ha^{-1} \cdot yr^{-1}$.
- Riparian wetlands with low anthropogenic N inputs have low N_2O emissions.
- Permanently flooded wetlands have lower N_2O emissions than those subject to fluctuating water tables.
- Riparian wetlands 'processing' high anthropogenic inputs of NO_3^- have significantly higher N_2O emissions than adjacent wetlands with low inputs.
- In riparian wetlands there is compelling evidence that NO_3^- is the main driver of N_2O emissions. The C:N ratio in wetland soil is a useful indicator of when significant N_2O production will occur. If C:N > 25, essentially zero N_2O is emitted.

Carbon sequestration in constructed treatment wetlands

There is a paucity of New Zealand data on C sequestration in wetlands in general, but particularly for marshes, including constructed wetlands, which are beginning to be installed for water quality enhancement, often following NIWA's construction guidelines

(Tanner et al. 2005). N_2O emissions are likely to be more of an issue for GHG emissions in these constructed wetlands because they are typically installed in high NO_3^- environments, with nitrogen attenuation by denitrification a key aim (Cooke et al. 2009). This raises the issue of 'pollutant swapping', whereby water quality may be improved at the expense of GHG emissions (Burgin et al. 2013). Nevertheless, some overseas studies (de Klein & van der Werf 2013; Mitsch et al. 2014) indicate that the net C sequestration of constructed wetlands is similar to, or greater than, that from natural flow through marshes.

A Waikato study investigated C accumulation (Tanner et al. 1998) and CH_4 emissions (Tanner et al. 1997) from pilot-scale constructed wetlands treating dairy shed wastewaters (i.e. higher C and nutrient loadings than for wetlands receiving farm run-off) under contrasting wastewater loading rates and wetland vegetation. The results from a low effluent loading rate in vegetated wetlands found the average CH_4 emission rate was 6.9 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$ and the organic matter accumulation rate was 6.7 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$, indicating close to zero net C sequestration.

Groh et al.(2015) researched GHG gas emissions (but not net sequestration) from two constructed wetlands treating agricultural tile drainage in Illinois and found that CO_2 was 75 and 96% of the total GHG emissions during two years. Most of the GHG came from terrestrial parts of the wetland and emissions increased with soil temperature. The fluxes of CH_4 and N_2O contributed between 2 and 11% and 3.7 and 13% of the total cumulative GHG flux, respectively, and the N_2O exchanges were 3.2 and 1.3% of the total nitrate removed from the two wetlands.

A recent review (Jahangir et al. 2016) of GHG and nutrient performance of constructed treatment wetlands (mainly treating domestic wastewaters) found that surface flow constructed wetlands decrease N_2O emissions but increase CH_4 emissions, whereas subsurface flow constructed wetlands increase N_2O and CO_2 emissions but decrease CH_4 emissions. Mixed species of vegetation perform better than monocultures in terms of increasing C and N removal and decreasing GHG emissions, but empirical evidence is still scarce.

The authors produced a conceptual model highlighting the current state of knowledge and experimental work that should be undertaken to address knowledge gaps across constructed wetlands, vegetation and wastewater types, hydraulic loading rates and regimes, and retention times, but there are a considerable number of information gaps. This review also highlights the methodological challenges of measuring net GHG emissions from constructed wetlands that are typically too small to use eddy covariance methods.

Potential for carbon sequestration in on-farm wetlands

There are few data for C sequestration on farm wetlands in New Zealand. Table A2 provides preliminary estimates of emission factors (EFs) for natural wetlands based on information from Ausseil et al. (2015) and Goodrich et al. (2017).

Table A 2: Preliminary estimates of emission factors for natural wetlands (marshes and mires). Constructed wetlands have no known data

Vegetation type	Age at maturity/steady state (where appropriate)	Maximum stock at maturity/steady state $t \cdot CO_2 \cdot ha^{-1}$ (Ausseil et al. 2015)	Stock assumed at clearance (if not max. stock) $t \cdot CO_2 \cdot ha^{-1}$	Time period assumed for sequestration	Mean sequestration rate $t \cdot CO_2 \cdot ha^{-1} \cdot yr^{-1}$	Refs
Wetland marshes (mineral)	Highly variable depending on setting	450	?*		?	nil
Wetland mires (peat)	Likely 10,000+ yr	5,000	NA	>50 yrs	2	
					0.78-3.55	Goodrich et al. 2017
					2.18-2.55	Campbell et al. 2014
					0.34	Schipper & McLeod 2002
Constructed wetlands	?	?	?		Insufficient NZ data	nil

* - ? = not known

The paucity of New Zealand data on C sequestration in both peat- and mineral soil-dominated wetlands coupled with the complexity of factors influencing the effects on the three dominant GHGs (CO_2, CH_4 and N_2O) in constructed wetlands result in *very high uncertainty* in estimating the potential for C sequestration in on-farm wetlands.

d) Proposed wetland classification

The current New Zealand UNFCCC wetland reporting classification has a single 'Vegetated Wetland' class. The wide variation in C stocks between mire and marsh wetland types (Table A1) and differences in the importance of emissions of the various GHGs between types (Figs. A3, A4) indicate that using this single class will result in high uncertainty in any CO_2e emission factor (EF) applied. Hence, development of a more granular classification to reduce uncertainty in EFs is desirable. However, this is very challenging due to the variety of wetlands that occur on farms and differences in GHG dynamics expected by wetland type, condition (e.g., level of disturbance by animal grazing and drainage), influent sediment and nitrogen loads, and position in the landscape (e.g., headwaters vs riverine floodplains). The next simplest classification might involve 3 classes: wetland remaining wetland; restored degraded wetland; and new land converted to wetland (i.e., constructed wetlands), ignoring wetlands degraded by grazing and drainage. However, further research on variations in C sequestration across wetland types is needed to develop and test a wetland typology or model to manage the uncertainty in assigning CO_2e emission factors to on-farm wetlands.

e) Research is needed to develop more comprehensive emission factors

There is a clear need for research to establish emission factors (EFs) for a wider range of New Zealand wetlands that occur on farms, and to understand the factors that influence the balance between CO_2 sequestration and CO_2e emissions of CH_4 and N_2O between wetland types so that generalised EFs can be developed with confidence. These need to include a temporal perspective, particularly for constructed and restored existing wetlands, because sequestration and emission are expected to change as systems become established and 'mature'.

There is a need for better information on the number, area and type of wetlands on farms. Resources such as farm riparian and environmental plans developed by Beef+LambNZ and regional councils and the DairyNZ Riparian Planner (https://www.dairynz.co.nz/environment/waterways/riparian-planner/) provide information at the scale required for dry-stock and dairy farms, respectively.

f) Wetland Summary

- The available New Zealand data on wetlands C show that they are a large store of C but have modest and highly uncertain annual C sequestration rates.
- There are two main types of natural wetlands – mires/peatlands and marshes/mineral soils – and a third new wetland class known as constructed wetlands (CW). Peatlands store six times more C than mineral soil wetlands. C storage in constructed wetlands is not known.

- Wetland CO_2 exchange is the balance of C sequestration and CH_4 and N_2O emissions, plus leaching of dissolved organic carbon to water drainage. The available New Zealand and some international data indicate wetlands are net CO_2 sinks. However, there is a paucity of local data in general, especially for mineral marshes and for constructed treatment wetlands designed for water quality protection, often in catchments with high NO_3^- nitrate loads and potential for emission of the powerful GHG N_2O from incomplete respiratory denitrification. This results in high uncertainty relating to the C sequestration from on-farm constructed wetlands and suggests this is an information gap requiring research.
- Ninety percent of New Zealand's original wetland area has been lost to farming, urban, and horticulture development, and continues to be lost.
- Wetlands provide multiple ecosystem services beyond C sequestration that currently benefit people and nature on farms and in catchments.

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B. Riparian strips (Larry Burrows, John Quinn & Elizabeth Graham)

a) Definition, characteristics, background and management

Establishment of riparian vegetation buffers of planted or naturally occurring grasses, shrubs and trees along waterway margins is becoming a widespread practice where farmland or production forests and waterways meet. *Riparian zones* are 'areas of direct interaction between land and water' (Figure B1). These interfaces are hotspots for important biogeochemical processes (such as denitrification, which converts nitrate (NO_3^-) to gaseous forms that are lost to the atmosphere) and ecological diversity.

How riparian land is managed has a disproportionately large influence on the aquatic environment of the area of land it occupies within the catchment, and this is exploited by *riparian buffers*, which are 'riparian areas that are managed to protect aquatic ecosystems from the effects of production or urban land uses'. There is some interaction and overlap between riparian buffers and wetlands, and a similar response by both to run-off inputs of nitrogenous fertilisers (see comments in the 'Wetlands' section above, and Cooke et al. 2009).

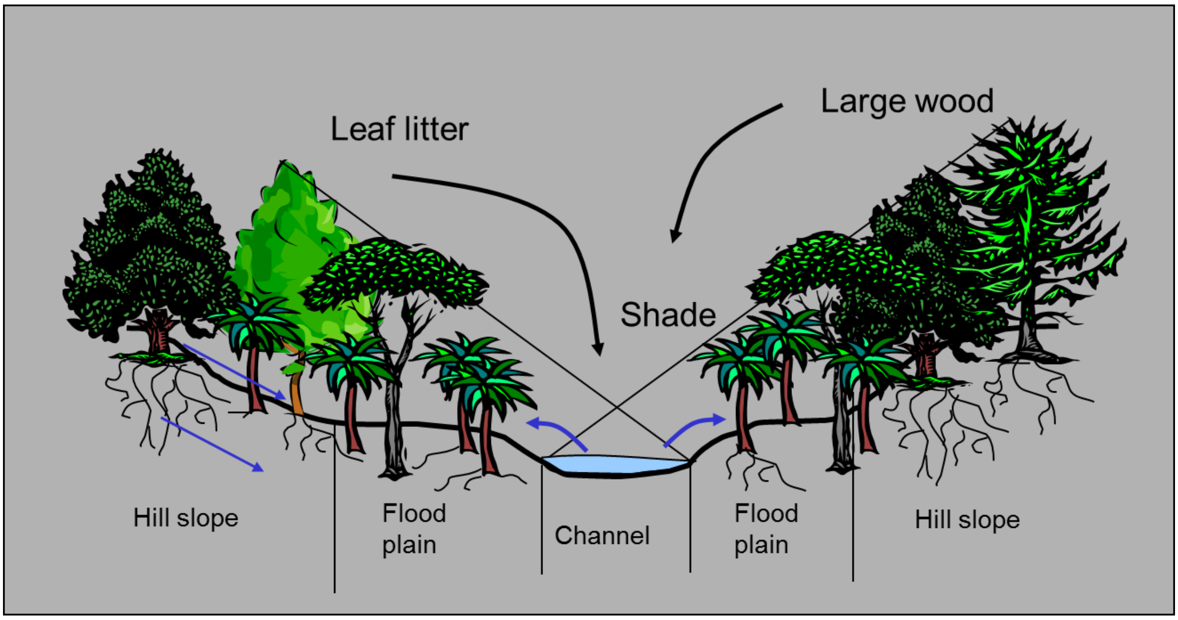

Figure B 1: Schematic of a stream riparian zone showing land–water C pools and influences including plants, flooding, shade, litter and wood input, and groundwater uptake. Copied from Parkyn (2004).

Actively established and managed riparian buffers have become widespread, particularly on dairy farms since the 2003 Dairying and Clean Streams Accord and the Sustainable Dairying: Water Accord (the Water Accord) in 2012. Taranaki Regional Council instigated the first regional riparian programme in 1993, resulting in 2,500 riparian plans (achieving 84% fencing and 70% planting) covering 14,500 km of streambank (https://www.trc.govt.nz/environment/freshwater/riparian-management/) by September 2017. Information on riparian conditions (fences, vegetation) is provided by Waikato Regional Council's 5-yearly on-the-ground surveys of dry-stock and dairy farms (Jones et al. 2015) and in plans collected by regional councils and DairyNZ (through their online riparian planning tool (https://www.dairynz.co.nz/environment/waterways/riparian-planner/). These sources indicate that typical riparian planted strip width on pastoral farms is c. 3 m.

In the Waikato region there was a significant increase in the proportion of stream length fenced on both sides of waterways between 2002 and 2012, resulting in an overall increase in the proportion fenced across the Waikato region from 34% to 51% (Jones et al. 2015). The increase has been greater along streams on dairy farms (25% increase to 70% in 2012) than on dry-stock farms (7% increase to 29% in 2012). This reflects the requirement of the Dairying and Clean Streams Accord for dairy farmers to have their 'Accord streams' – those wider than a stride (or c. 1m) and deeper than a 'Redband' (0.3 m) – to be fenced by 2013. Over the 5 years to 2012 the rate of change was about 3.5% of bank length per year for dairy farms and about 0.2% of bank length per year for dry-stock farms.

The ability of riparian vegetation to buffer over-ground runoff, below-ground flows of particulates and nutrients, logging debris from plantations while shading the aquatic ecosystem is affected by their width, height of the vegetation and the slope of the ground (Parkyn 2004). Different setback widths are required to buffer different contaminants, or if self-sustaining natural regeneration is expected within the strip. Planting setbacks from streams adjacent to production plantation forests (as required by the National Environmental Standard for Plantation Forestry: i.e. within 5 m of a perennial river with a bankfull channel width less than 3 m, or within 10 m of a perennial river with a bankfull channel width of 3 m or more) to encourage natural regeneration of riparian vegetation that can provide substantial protection of stream habitat and biota during the harvest phase if left relatively undisturbed (Quinn 2005).

Along the farmed margins of large (often braided) rivers throughout New Zealand, riparian vegetation commonly consists of extensive areas of woody weeds such as gorse, scotch broom and willow trees. In places mixtures of woody weeds and regenerating native shrub and tree species occur. Management of these areas tends to be either low input, or intensive weed control to recover areas of river terrace for pasture. Such areas could be managed for C sequestration under existing Emissions Trading Scheme (ETS) rules as they mostly exceed 30 m in width but would need to have sufficient trees species established to meet future forest cover and height rules (see Burrows et al. 2015). These areas mostly fit within existing ETS offsetting rules so are not considered further here.

The main purposes for establishing riparian buffers are to protect stream habitat and lessen the effects of agricultural and forestry spill-over on water quality and aquatic ecosystems. There are numerous ancillary co-benefits, however, including:

- excluding stock from streams, hence reducing nutrient and pathogens from dung
- reducing sedimentation from bank erosion
- protecting streambanks from fluvial erosion and livestock treading damage
- filtering particulate nutrients, pathogens and sediment in surface run-off
- uptake and denitrification of nutrients in shallow groundwater
- providing habitat and shading for both terrestrial and aquatic biota

Recent results by Graham et al. (2018) show that the Taranaki Riparian Management Programme, which began in the early 1990s, is beginning to show measurable beneficial effects on water quality and downstream aquatic invertebrate communities.

Results from farmer surveys on the pros and cons of riparian planting have not raised C sequestration as a co-benefit (Maseyk et al. 2017), although others have suggested the possibility (Shepherd et al. 2017). Riparian buffers can also benefit agro-ecosystems by providing shelter for livestock during extreme weather events, habitat for pollinators, honey, timber, etc. (White et al. 2014; Maseyk et al. 2017). Indications are that riparian restoration can result in a net cost–benefit to farms if all beneficial services are taken into account, as well as creating a GHG offset sink (Daigneault et al. 2017; Shepherd et al. 2017).

Riparian establishment in strips or belts of new vegetation that are taller and woodier than adjacent pasture, especially where shrubs or trees are present, can be reasonably assumed to sequester biomass C. Grass-only riparian strips are beneficial for buffering above-ground sediment transport (Parkyn 2004). Riparian restoration plan guides recommend establishing belts of flaxes, grasses, sedges/rushes, shrubs and trees, depending on regional location (https://www.dairynz.co.nz/environment/waterways/planting-waterways/#guides), rather than just a narrow stand of forest trees. These guides plan for flooding inundation along the water edge, with shorter, water-tolerant vegetation in a lower-bank zone, a central upper-bank zone of taller trees and shrubs, and a grass strip buffer along the setback pasture boundary maintained by grazing stock and resulting in a characteristic vegetation structure along stream banks.

This structure specific to riparian strips, also suggests there will likely be a different biomass profile from other successional shrublands or grassland land-cover classes. This has implications for vegetation growth, total biomass and C sequestration, and suggests that existing models of sequestration for forest succession (e.g. MPI indigenous forest look-up tables (https://www.mpi.govt.nz/.../6979-look-up-tables-for-post-1989-forest-land-in-the-ets) are inadequate representatives of sequestration in riparian strips. Age to maturity, when sequestration tapers off, is also unknown, as are disturbance rates from, for example, flooding events.

The amount of C sequestered will simply depend on the biomass (plant size per volume and mass dimensions) of the riparian vegetation, its rate of growth over time, and the area established. The width of typical riparian restoration strips varies from 1 to a few tens of metres or more, and due to its linear nature that width has a major effect on total hectares of riparian vegetation, and hence sequestered C on any farm; for example, a 3 m wide strip of 1 ha is c. 3.3 km long.

Preliminary information from the DairyNZ / Manaaki Whenua Riparian Planner (https://riparian-planner.dairynz.co.nz/) indicates that nationally, across all dairy farms with streams to manage, a median stream length of 3.0–3.1 km with a median setback on either side of 3 m is being actively planned for by dairy farmers (Tom Stephens, pers. comm.; based on c. 408 dairy-specific plans, filtered to exclude likely scenario plans from the total Riparian Planner database).

b) Emission factor literature review

There are no known quantitative data for C sequestration or EFs for planted riparian strips in New Zealand, although there are data for shrublands dominated by woody weeds such as scotch broom and/or gorse (Carswell et al. 2013, 2014; Easdale et al. 2015). Daigneault et al. (2017) used results derived from national GHG inventory methodologies (MfE 2017) where riparian plantings are included as 'Grassland with Woody Biomass – Transitional', and where an emission factor of –0.43 $t{\cdot}C{\cdot}ha^{-1}yr^{-1}$ and mature stock of 28 $t{\cdot}C{\cdot}ha^{-1}$ was used. Others have used the indigenous forest Look-up tables (MPI, https://www.mpi.govt.nz/growing-and-harvesting/forestry/forestry-in-the-emissions-trading-scheme/) (see Taranaki Ring Plain example in section G. below).

For the purpose of this exercise, and to provide indicative sequestration rates, riparian strips are presumed to be similar to early successional shrublands (Carswell et al. 2013, 2014; Holdaway et al. 2010, 2017) rather than plantations of forest trees or successional indigenous forest (MPI look-up tables), or 'Grassland with Woody Biomass' (MfE 2017). All these sources of vegetation sequestration rates are surrogates for unbiased riparian-specific rates.

c) Current classification for NZ UNFCCC reporting and accounting

Riparian strips in New Zealand do not have their own emissions profile. In the UNFCCC, New Zealand's Greenhouse Gas Inventory 1990–2015 (MfE 2017), riparian strips, where identified, were included as 'Grassland with Woody Biomass – Transitional'. A sequestration rate of 0.47 $t{\cdot}C{\cdot}ha^{-1}$ (1.71 $t{\cdot}CO2{\cdot}ha^{-1}$) and a C stock maturity (28 yrs) of 13.05 $t{\cdot}C{\cdot}ha^{-1}$ (47.85 $t{\cdot}CO2{\cdot}ha^{-1}$) are used, based on a Land Use and Carbon Analysis System (LUCAS) plot-based estimate for that class by Wakelin & Beets (unpubl).

d) Proposed emission factors for New Zealand and data

All the following EFs are for exotic shrublands and/or successional shrublands. It is not known how they compare with data from actual riparian restoration sites. For reference, existing sequestration rates for various successional shrublands follow.

- Thirty-year-old gorse stands are generally around 4 m in height and contain approximately 125 $t{\cdot}C{\cdot}ha^{-1}$ (458 $t{\cdot}CO_2e{\cdot}ha^{-1}$). This equates to c. 4.1 t·C $ha^{-1}{\cdot}yr^{-1}$ (15.3 $t{\cdot}CO2{\cdot}ha^{-1}{\cdot}yr^{-1}$) t·C (Carswell et al. 2013).
- Initial calculations give 1.24 t·C (4.6 ± 0.2 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$) for actively regenerating kānuka forest and tall shrubland and mean estimates of 0.72 t·C (2.7 ± 0.07 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$) for regenerating forests (Holdaway et al. 2010).
- Empirical estimates give 1.42 t·C (5.28 ± 0.99 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$) based on a comprehensive sample of 104 re-measured early successional woody communities (Carswell et al. 2014).

Shrubland sequestration rates vary according to age, and also with environmental conditions and species, so any comparison (such as the values given above) should ideally take these other variables into account. Holdaway et al. (2010) also indicate considerable differences between regions: Nelson (4.1 ± 0.7 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$), Northland (3.7 ± 0.3

$t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$), Gisborne (3.7 ± 0.4 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$), and Manawatū/Wanganui (3.6 ± 0.3 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$), whereas the slowest correspond to the West Coast (1.7 ± 0.2 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$) and Otago (2.1 ± 0.3 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$).

Table B 1: Emissions factors for riparian strips derived from various shrubland data

Vegetation type	Age at maturity/ steady state (where appropriate)	Max. stock at maturity/ steady state $t{\cdot}CO_2{\cdot}ha^{-1}$	Time period assumed for sequestration	Mean sequestration rate $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}\ yr^{-1}$	Source(s)
Other land	NA	NA	NA	0	MfE 2017
Grassland w woody biomass	28	47.85	28	1.7	Wakelin & Beets unpubl.
Shrublands				5.28	Carswell et al. 2014
Gorse	30	458	30	15.3	Carswell et al. 2013
Broom/tauhinu shrubland		43/87		??	Easdale et al. 2015

e) Riparian Summary

- In lieu of specific riparian EFs, examples of sequestration rates from New Zealand shrublands have been given here, although they should be treated with caution due to the factors mentioned above.
- Estimated median surrogate values for sequestration are c. 3.4 $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}\ yr^{-1}$.
- Establishment of planted riparian strips on farm stream boundaries is increasing rapidly in New Zealand as the result of national initiatives such as the Dairying and Clean Streams Accord and the Sustainable Dairying: Water Accord (the Water Accord).
- Measurable beneficial effects on water quality and downstream aquatic invertebrate communities are now being attributed to riparian strips and riparian management programmes that have been in place for over 2 decades.
- Vegetation in riparian strips includes grasses, sedges, shrubs, flaxes and trees in a profile bound by water on one side and a fence on the other. They cannot be equated with forests when considering C sequestration and stocks. Existing models of sequestration for forest succession (e.g. MPI indigenous forest look-up tables) are inadequate representatives of sequestration in riparian strips. Age to maturity, when sequestration tapers off, is also unknown, as are disturbance rates from, for example, flooding events.
- Developing specific EFs for New Zealand riparian strips requires a focused research programme that would sample vegetation and soils for a range of chronosequence riparian strips across a selection of environments or regions that account for specific stand structure, stand age, environmental conditions, and species mixes.

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C. Pole plantings (Alec Mackay)

a) Definition, characteristics and management

Poplars (*Populus* spp.) and willow (*Salix* spp.) are exotic fast-growing species ideally suited to both prevent and control erosion, due to their extensive and deep root network. Plantings of poplar and willow as poles on farms have been used to stabilise gully, earthflow and slope erosion in New Zealand since the 1950s. Hill land (slopes >15°) covers 10 million ha of the country, often prone to slight to moderate erosion (Lynn et al. 2009), with almost 20% of hill land prone to severe or extreme erosion.

Thousands of hectares of wide-spaced poplar (*Populus* spp.) and willow (*Salix* spp.) trees are planted in hill pastures annually to reduce the risk of erosion and protect natural and built capital improvements and the flow of a wide range of services (Wilkinson 1999; Douglas et al. 2013; Dominati et al. 2014). The biggest threat to C stocks in hill land is erosion (Lambert et al. 1984; Douglas et al. 1986). Other species such as wattles (*Acacia* spp.), gums (*Eucalyptus* spp.) and alders (*Alnus* spp.) are planted at wide-spacings for soil conservation, but they are used less frequently (Van Kraayenoord & Hathaway 1986).

Figure C 1: Widely space-planted poplar and willow.

The advantages of poplars and willows over other species are that both can be planted as poles and are best suited to sites where erosion is mediated by surplus soil water. Trees established well and maintained effectively enable the continuation of pastoral livestock enterprises on land that otherwise requires the removal of livestock either to allow the planting of exotic or native plantation forestry or to facilitate indigenous forest regeneration. In addition to erosion control they are used in shelter belts, and provide shade and shelter, drought fodder, and a timber resource. Willow plantings can also be used to help stabilise stream banks, as their fibrous root systems help knit and bind the banks. Poplars should be planted a little further back from the edge of the bank.

Poles are planted in the winter and protected with a plastic sleeve to increase the survival rate of the poles and make it difficult for possums to climb them. The sleeves also protect the poles from stock damage and reduce moisture loss. The young trees are form pruned at 3–5 and 7–10 years to limit branching and shading.

In some parts of the country poplar and willow are an important source of fodder for livestock, particularly during the summer months in dry environments (MacGregor et al., 1999).

While the primary purpose of the trees is to reduce the occurrence of soil erosion, there is increasing interest in the potential of trees to sequester C in above- and below-ground components to mitigate increasing atmospheric concentrations of greenhouse gases.

Wide spaced planted conservation trees

The stem density of mature, wide-spaced trees on pastoral hill country is usually less than 50 stems per hectare (Hawley & Dymond 1988), equivalent to 14 m spacing, although density can vary considerably depending on the type and severity of erosion.

Douglas et al. (2009) reported that wide-space planted trees at densities of 30–60 stems·ha^{-1} (13–18 m spacing) reduced soil slippage at 65 sites by an average of 95%. It was suggested that trees under about 10 cm in diameter (DBH) provide little or no benefit, while for trees over 30 cm DBH, the benefit from 100 stems·ha^{-1} is no greater than from 36 stems·ha^{-1}. McIvor (2015) recommends a spacing of 12–15 m on slopes, with closer spacing where the slope is unstable. About 30–50 mature stems·ha^{-1} (18–14 m spacing) was considered necessary for water management, topsoil retention and slope protection over erosion-prone land, while enabling 87–92% of pasture production of an open pasture.

In a modelling exercise, Vibart et al. (2015) explored the impact of wide-spaced conservation tree plants on the GHG profile of the farm as it influenced understorey pasture growth, livestock performance, and feed supply. They suggested that the shelter benefits from ewes having free access to the blocks containing space-planted trees were captured in reduced lamb losses relative to an open pasture, and total GHG emissions reduced from 4.8 t·CO_2e·ha^{-1}·yr^{-1}in open pasture to 4.2 – 4.4 t·CO_2e·ha^{-1}·yr^{-1} in the blocks containing spaced trees.

Current situation

Approximately 130–150,000 poles were planted in 2017, of which 60–70% were poplar and the balance willow. Pole planting in the Horizons, Hawke's Bay, Greater Wellington, and Gisborne District Council regions made up the bulk of plantings. On the East Coast more willow were planted than poplar. On average, a land-owner plants 100–120 poles in any one year.

The numbers of poles being planted are increasing, with nurseries looking to build their capacity to produce poles. For example, Northland is one region looking to increase the number of poles planted.

b) Emission factor literature review

There are a large number of published studies (e.g. Douglas et al. 2009, 2018) and a number of unpublished data (Ian McIvor, Plant and Food Research, pers. comm.) on poplar tree growth rates in New Zealand. Results from both structured experiments (e.g. Nelder radial planting studies; Nelder 1962) and field-based sampling studies at a range of densities could be used to develop a good understanding of C sequestration rates of poplar stands on farm land from the time of planting through to maturity, between 40 and 60 years, as influenced by climate, slope and soil type. In comparison to poplar there are very few data sets for willow. To address this knowledge gap a field-based programme of research would be required.

Although there are studies of tree dimensions and growth, few estimates of carbon stocks or sequestration rates have been reported for widely spaced poplars. Guevara-Escobar et al. (2002) found the total agroecosystem carbon measured in a mature poplar-pasture system (55.5 $tC{\cdot}ha^{-1}$ (203.5 $t{\cdot}CO_2e{\cdot}yr^{-1}$)) was 26% higher than in an open pasture system without trees (44.0 $t.C{\cdot}ha^{-1}$ (161.3 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$), with the extra carbon residing in the poplar trees. The sequestration rate in the trees over 30 years (37 stems per ha, or 16 m spacing) was approx. 2 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$. Data from Douglas et al. (2018) suggest poplar growth rates are not affected by planted densities in hill country, indicating the values reported by Guevara-Escobar et al. (2002) could be scaled up or down with a change in stems per ha. For example, at 100 stems per ha C sequestration would more than double per hectare until the stand reaches maturity.

There is a question about the effect of wide-spaced trees on soil C. Douglas et al. (2018) examined the influence of space planted conservation trees on soil C and found that under poplar there was less soil C than in adjacent open pasture. This suggests more work needs to be done to clarify the net C balance of wide-spaced conservation plantings compared with open pasture before developing New Zealand-specific emissions factors.

Carbon stocks and sequestration will often be constrained by management. It is important to remember the objective of soil conservation planting is to minimise the number of stems per ha and to minimise the canopy cover of each tree through the breeding of clones with a narrow form and form-pruning of trees in the first 2–10 years. Coppicing of trees to manage size is practised, particularly in steep hill country, to limit the loadings on hill slopes, while keeping the tree standing biomass down in areas with high wind runs to reduce the risk of toppling.

c) Current classification for NZ UNFCCC reporting and accounting

Wide-spaced conservation plantings are currently included as forest if they meet the 30% cover threshold and other criteria of ETS forests for area, height and width. In that case they would be eligible for the ETS and for areas less than 100 ha could use the MPI Lookup table for Hardwoods, although actual sequestration would be proportionally less compared with fully stocked stands. If the canopy cover threshold isn't met they would just be mapped as grassland (high or low producing).

d) Proposed EFs for NZ and data

Little data exist for sequestration in New Zealand wide-spaced conservation plantings, and as mentioned above, further research is necessary to identify net growth and sequestration rates and achievable carbon stocks at maturity. With active management of the tree (e.g. pruning, coppicing, pollarding) to optimise their use in a pastoral system, further research is required to better understand the influence this has on C sequestration.

Table C 1: Sequestration rates for Pole Planting in New Zealand

Vegetation type	Maximum stock at maturity/ steady state $t.CO_2{\cdot}ha^{-1}$	Stock assumed at time of clearance (if not max stock) $t.CO_2{\cdot}ha^{-1}$	Time period assumed for sequestration	Mean sequestration rate t·CO2e·ha−1·yr−1	Source(s)
Wide-spaced Pole Plantings				4.38 *	*ETS Hardwoods lookup table*
Wide-spaced Poplars	42.2	37 stems·ha^{-1}	30	2.0	Guevara-Escobar et al. 2002

*This value in the lookup table is for fully stocked stands, and so is reduced proportionally from 300 to 50 stems per ha.

e) Pole Planting Summary

- Widespread on hill country farms.
- Numbers of poplar and willow poles being planted is increasing.
- Prime purpose of poplar and willow pole planting is to reduce the risk of erosion to protect natural capital and soil carbon stocks.
- Increasing sequestration is partly in conflict with the management of the trees to maximize effectiveness for reducing erosion while minimising their impact on pasture growth.
- There are some data on poplar clones and their growth rates, but no data for willows.
- There is a scarcity of data on sequestration rates for both poplar and willow.
- Requires a research programme to resolve EFs that account for species, soils, national regional differences, age to maturity, and decide on a baseline.

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D. Shelterbelts (Steve Wakelin)

a) Definition, characteristics and management

Although the climate in New Zealand is generally mild enough for livestock to graze outside all year, there are still benefits to both pasture and livestock from shelter and shade. Vegetation planted for shelter on New Zealand farms includes hedges and shelterbelts. A hedge is usually a dense, single row of a low-growing shrubby species, or a trimmed and topped tree species. Originally gorse was often used, but now it is common to plant native species for low shelter, including non-woody species such as toetoe (*Austroderia* spp.) or harakeke (flax, *Phormium tenax*). Boxthorn hedges (*Lycium ferocissimum*) are a feature in Taranaki dairy farms but are now classified as a pest-plant (weed). Shelterbelts are typically taller hedges of tree species, but are also often mechanically trimmed and may be topped. Hedges and shelterbelts are particularly common around farm dwellings and horticultural crops and in areas subject to cold or drying winds.

Old shelterbelts are often single rows of radiata pine or macrocarpa, but the inclusion of a second, slower-growing row has been widely promoted. The aim is to maintain denser shelter lower to the ground and about 40% porosity above that to reduce turbulence. Effective shelter may be provided for a distance downwind that is 20–30 times the height of the trees (Figure D1).

In practice shelterbelts and hedges on New Zealand farms are highly diverse. This is in part because of the specific topography and layout of farm paddocks, but also because there are multiple drivers for the establishment of shelter and also some negative impacts or barriers to consider (Table D1). Diversity arises because of the compromises that are made.

Table D 1: Drivers and barriers for shelterbelt establishment

Drivers / positive impacts of shelter	Barriers /negative impacts of shelter
Improve livestock welfare and productivity, including young stock survival and growth	Cost of fencing, establishment and maintenance
Increase pasture productivity	Loss of production land
Improve irrigation efficiency from reduced evaporation	Uncertain benefits
Reduce wind erosion of topsoil	Time required
Provide habitat for wildlife	Impede modern large-scale irrigation
Create a more diverse, pleasant working and living environment	Compete with pasture for water and nutrients
Reduce crop damage	Creation of frost pockets
Provide stock fodder	Shade tracks, preventing them from drying out or thawing
	Impede line of sight
	Boundary disputes, power lines, TranzRail, civil aviation regulations, public roads
	Damage to fences from falling branches
	Harbouring pests and weeds
	Greater wind turbulence beyond the shelter influence
	Wilding spread potential

Specialised variations may include:

- timber belts – one or two rows of pruned radiata pine intended for producing pruned logs, usually with a row of a slower-growing species; however, the wood quality from shelterbelts is generally poor, so timber belts have largely fallen from favour, and wood from felled shelterbelts is more usually used for firewood
- stock havens –several rows of trees, effectively forming a small woodlot where stock can find shelter in extreme weather conditions.

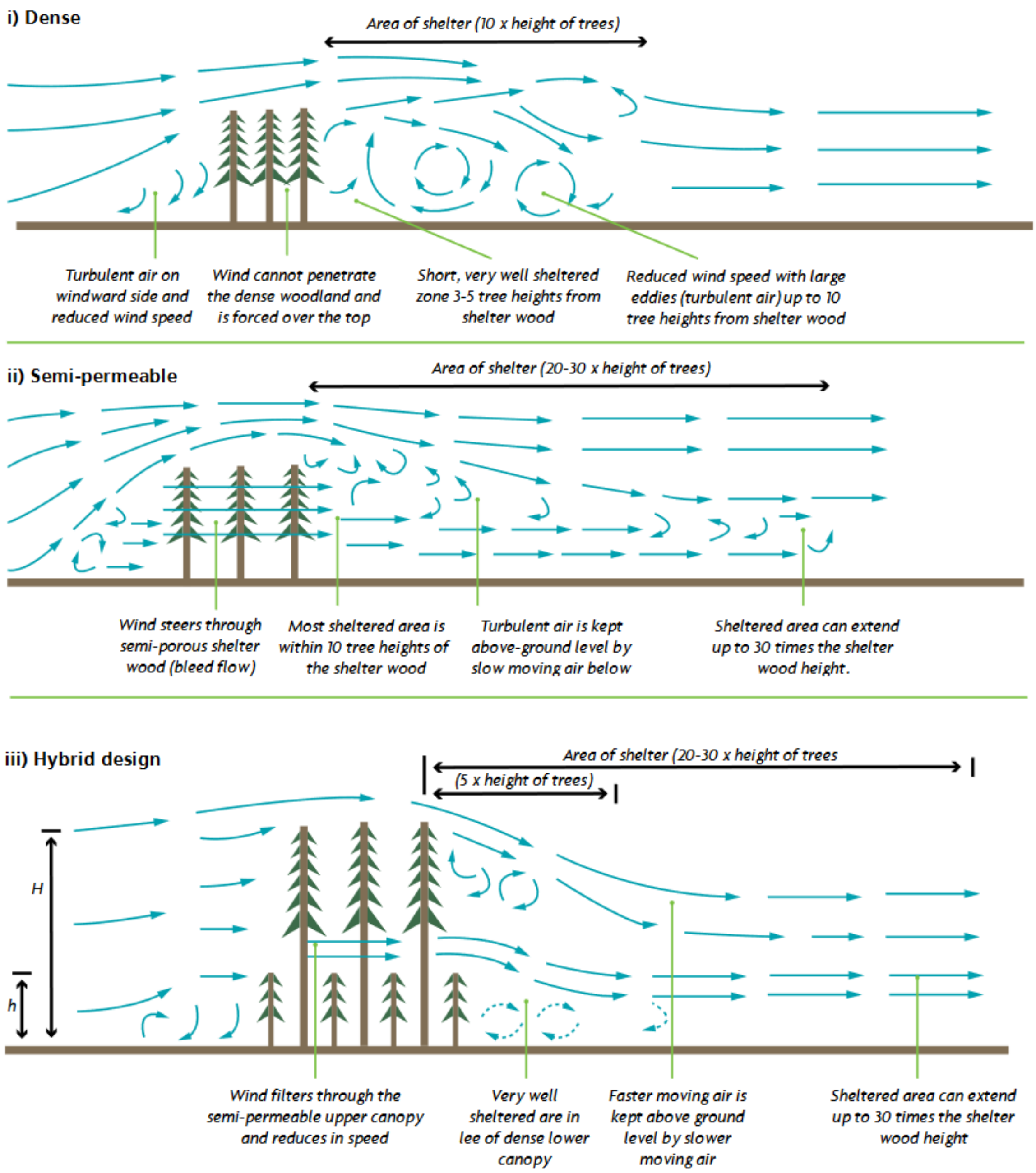

Figure D 1: Types of shelterbelts and effects on air flow.
Source: DairyNZ (https://www.dairynz.co.nz/media/5447838/Trees_for_shelter.pdf)

In horticulture the density of shelterbelts is much higher than in grasslands. The main species planted in the past have been *Cryptomeria*, *Casuarina*, willows (*Salix*) and poplars (*Populus*). Alders (*Alnus*) have since become more popular. Artificial shelter (polyethylene fabric attached to tall posts) is often preferred for vineyards because trees encroach on productive space, restrict air movement and provide roosting sites for pest bird species (Jackson & Schuster 1997).

Willows, eucalypts and poplars chosen as fast-growing shelter species for crops may compete too vigorously and need removal and replacement after 20–30 years. Shelterbelts of radiata pine and macrocarpa in grasslands can reach very old ages, but many have been removed to allow irrigators to pass. *Miscanthus* (a toetoe-like grass grown for bioenergy) has been suggested as an alternative low shelter species on the Canterbury plains for this reason.

b) Classification for NZ UNFCCC reporting and accounting

Shelterbelts of tree species capable of reaching 5 m in height that are wider than 30 m and more than 1 ha in size qualify as forests in New Zealand's GHG inventory and under the ETS. Hedges are generally too short as well as not wide enough, while shelterbelts are typically of one to three rows and will not reach the width threshold, especially if trimmed.

In the national GHG inventory, linear shelterbelts are included variously in the mapped area of:

- annual cropland
- perennial cropland
- high-producing cropland (if <1 ha and <30 m)
- low-producing cropland (if <1 ha and <30m)
- grassland with woody biomass (GWB) (if <1 ha and <30 m)
- settlements (`grassland within urban settings including … urban parklands and open spaces that do not meet the forest definition').

According to the LUCAS satellite imagery interpretation guide (MfE 2012, p. 55), a shelterbelt may be large enough to be mapped and assigned to 'Grassland with Woody Biomass'. In practice this does not happen: a shelterbelt of over 1 ha that is >30 m in width would be mapped as forest; smaller shelterbelts would be mapped as the predominant surrounding land use.

New Zealand's emission factor for perennial cropland is a weighted average, taking into account grapevines, kiwifruit vines, fruit trees and shelterbelts. Perennial cropland trees that are primarily grown for fruit or nuts are not considered forests under the ETS or in the GHG inventory. A relatively high proportion of the biomass in land mapped as perennial cropland is contained in the shelterbelts (48–87% depending on the crop, see Table D2). If these shelterbelts are to be mapped separately, the EF for perennial cropland will need to be revised.

The EFs used for annual cropland, high-producing grassland and low-producing grassland in New Zealand's GHG inventory are all IPCC defaults that do not include a contribution from shelter vegetation. The GWB EFs used in the inventory are based on shrubland communities, with no representation from shelterbelts.

c) Emission factor literature review

EFs are required to reflect the C sequestration rate of growing shelterbelts and the emissions resulting from shelterbelt removal. The latter may be the C stock at maturity/steady state, or the stock at the assumed removal age. If the age of removal is unknown, it is appropriate to use a time-averaged C as the basis for estimating emissions i.e. the average stock from the time of shelterbelt establishment until its removal. This may be approximately equal to the final stock divided by two (assuming a constant sequestration rate) but will be closer to the final C stock value itself if the shelterbelt reaches a steady-state stock at maturity years or decades before removal.

There are currently no IPCC default EFs for hedges or shelterbelts. Estimates in the literature are usually based on modelling or allometric equations rather than destructive biomass sampling. Care must be taken to distinguish between shelterbelt C stock estimates that are expressed on a 'per hectare of cropland or grassland' basis (i.e. taking into account the proportions of shelterbelt and non-shelterbelt area in the land use) and those that are expressed 'per hectare of shelterbelt' (i.e. taking into account only the area occupied by the shelterbelt).

Kerckhoffs and Reid (2007) estimated C stocks in perennial crops in New Zealand and included an estimate for the C in shelterbelts based on Canadian work (Kort & Turnock 1999). The Canadian study included a range of species grown as shelterbelts on Canadian prairies. Most of the sampled shelterbelts were mature, aged 30–60. Results for above-ground C stocks were:

- 18 t dry mass km^{-1} (18 $t{\cdot}dm{\cdot}km^{-1}$) for shrubs (mean of five species, range 11–26 $t{\cdot}km^{-1}$)
- 31 $t{\cdot}dm{\cdot}km^{-1}$ for conifers (mean of three species, range 24–41 $t{\cdot}km^{-1}$)
- 52 $t{\cdot}dm{\cdot}km^{-1}$ for deciduous species (mean of four species, range 31–104 $t{\cdot}km^{-1}$)
- 104 $t{\cdot}dm{\cdot}km^{-1}$ for improved, fast-growing poplar (*Populus x deltoides*) clones.

The poplars were harvested at an average age of 33 years and are probably the most relevant of the species examined to a New Zealand orchard situation. Kerckhoffs and Reid (2007) assumed an orchard shelterbelt length of 400 $m{\cdot}ha^{-1}$, so the Canadian poplar estimates give a C sequestration rate of 2.31 $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}yr^{-1}$ and a stock at maturity of 76.3 $t{\cdot}CO_2{\cdot}ha^{-1}$ of cropland.[6] However, the authors noted that the biomass stock figures for shelterbelts used on the Canadian prairies are probably more than double the estimates that might be expected under New Zealand orchard situations where the trees are regularly trimmed. Thus annual C fixation by shelterbelts in an orchard situation may be more in the order of 1.15 $t{\cdot}CO_2{\cdot}ha^{-1}$ of cropland.

[6] That is, [104 $t{\cdot}dm{\cdot}km^{-1}$ * 0.5 (carbon fraction) * 0.4 (km^{-1}) * 11/3] / 33 years

Tombleson (1986) measured stem volume in a young *Eucalyptus nitens* firewood belt at Reporoa, near Taupō, which was felled at age 9 years. The trees had been planted in two rows spaced 3 m apart with 2 m between trees. Total stem volume from 770 trees per kilometre was 265 m^3, or 0.34 m^3 per tree. Assuming a wood basic density of 433 $kg \cdot m^{-3}$ (Tombleson 1986), wood mass amounted to 149 kg per tree, or 74 $t \cdot km^{-1}$ at 2 m spacing. Assuming an orchard situation with 400 $m \cdot ha^{-1}$ of shelterbelt (and a C fraction of 0.5), C sequestration would amount to 9.2 $t \cdot CO_2 \cdot ha^{-1} \cdot yr^{-1}$ in the stem wood alone over 9 years, a rate four times greater than for unpruned poplars in Canada.

The EF for perennial cropland currently used in New Zealand's GHG inventory was derived by Davis and Wakelin (2010). They followed Kerckhoffs and Reid (2007) in assuming that half of orchards would be sheltered, based on the fact that although shelterbelts are almost always planted around kiwifruit orchards, they are used less commonly around pip fruit orchards and rarely around vineyards (which make up 40% of perennial cropland area).

Table D 2: Indicative cropland factors (values are per hectare of cropland)

Crop	Biomass accumulation rate ($t \cdot dm \cdot ha^{-1} \cdot yr^{-1}$)	Carbon accumulation rate ($t \cdot C \cdot ha^{-1} \cdot yr^{-1}$)	Harvest/ maturity cycle (yr)	Above-ground biomass carbon stock at harvest ($t \cdot C \cdot ha^{-1}$)
Kiwifruit	0.6–0.91	0.3–0.46 (0.38)	40–45	12–20.5
Grapes	0.38–0.87	0.19–0.44 (0.31)	20	3.8–8.7
Pip fruit	0.38–0.96	0.19–0.48 (0.33)	15–20	2.8–9.6
Shelter belt	1.26	0.63	30	18.9
NZ combined perennial cropland (assumes half of crop area has shelterbelts)		0.67 (2.5 $t \cdot CO_2$)	28	18.8 (68.9 $t \cdot CO_2$)

More recent work has investigated C stocks in shelterbelts on the Canterbury plains. Czerepowicz et al. (2012) estimated the C stocks in conifer shelterbelts using multispectral imagery and sampling for above ground biomass (AGB), based on measured diameters and heights and the allometric equations of Moore (2010). Shelterbelts were estimated to contain on average 869 ± 282 $t \cdot CO_2e \cdot ha^{-1}$ of shelterbelt. Given the density of shelterbelts in the region, this was estimated to be equivalent to 22 $t \cdot CO_2 \cdot ha^{-1}$ of grassland.[7]

[7] The authors stated that coniferous shelterbelts contribute up to an extra 6 $t \cdot C \cdot ha^{-1}$ to the 'grassland with woody biomass' carbon pool in New Zealand, which was given as 29 $t \cdot C \cdot ha^{-1}$. This interpretation is not correct (and the 29 $t \cdot C \cdot ha^{-1}$ GWB estimate has been superseded). If the shelterbelt sampling was representative of the landscape, no shelterbelts were mapped as forest or GWB, and all of the area was mapped as high-producing grassland, then the EF for high-producing grassland in that region could be increased by about 6 $t \cdot C \cdot ha^{-1}$ (i.e. 22 $t \cdot CO_2 \cdot ha^{-1}$).

Welsch et al. (2016) quantified shelterbelt C stocks on an organic mixed cropping farm and a conventional dairy farm in Canterbury (Table D3). The shelterbelts had been established for over 30 years and included poplar, radiata pine, macrocarpa, oak, and natives (flax, cabbage tree, *Pittosporum*). Shelterbelts were trimmed every 1–2 years. Above-ground biomass was again estimated for radiata pine based on Moore's (2010) allometric equations. A Canadian equation was used for poplar and an Austrian equation for oak. These equations gave very high C stock estimates (Table D3). The native shelterbelts were only 4–5 years old so provide limited useful information.

Table D 3: Carbon stock estimates (per hectare of shelterbelt) for shelterbelts at Lincoln, Canterbury

Species/Farm type	Age	Height (m)	Width (m)	AGB $t{\cdot}CO_2{\cdot}ha^{-1}$	CWD $t{\cdot}CO_2{\cdot}ha^{-1}$	Litter $t{\cdot}CO_2{\cdot}ha^{-1}$
Macrocarpa – dairy	30	8	5	116.2	123.6	5.5
Macrocarpa – organic	25	4.5	5	93.9	79.6	7.0
Natives – dairy	4–5	2.3	3.3	8.1	0.0	4.8
Natives – organic	4–5	2.3	3.3	12.1	0.0	4.0
Radiata – dairy	30	8	4	129.1	211.6	23.5
Oak – organic	35	18.5	5	1,223.9	205.3	9.9
Poplar – dairy	30	24	1.5	1,927.2	41.8	1.1
Poplar – organic	30	24	1.5	2,369.8	51.7	2.9

Source: Welsch et al. 2016

The work described in Welsch et al. (2016) was a pilot project for a PhD thesis (Welsch 2016), which included more detail on 59 shelterbelts in Canterbury (Table D4) and focused on soil processes.

Table D 4: Shelterbelt dimensions for native and Radiata pine in the Welsch 2016 study

	Native shelterbelts	Radiata pine shelterbelts
Number of shelterbelts	25	34
Average age (range)	16 (6–50)	27 (15–45)
Dbh[+] (range) cm	(6–31)	(23–159)
Height (range) m	(2–20)	(5–25)
Number of rows (range)	(1–20)	(1–20)
Average spacing	1.9 m between trees; 39 trees/shrubs per 100 m	2.1 m between trees; 60 trees (all rows) per 100 m
Total carbon (AGB[+] + litter + BGB[+] + soil) Mean $t{\cdot}CO_2{\cdot}ha^{-1}$ ± std error	406.3 ± 469.3	559.5 ± 480.3
Biomass and litter $t{\cdot}CO_2{\cdot}ha^{-1}$ (c. 62%)	251.9	346.9
Soil carbon (c. 33%) to 10 cm$\cdot t{\cdot}CO_2{\cdot}ha^{-1}$	134.2	184.8

[+]DBH – Diameter at breast height, AGB – Above ground biomass, BGB – Below ground biomass

While the study concluded that native shelterbelts, in particular, have considerable potential for increasing C pools, there are limitations to the use of this work to provide default emissions factors.

- It provides a snapshot in time of current shelterbelt stocks rather than a basis for modelling shelterbelt C stocks by age. Stocks for each shelterbelt are not presented, so trends with age could not be assessed.[8]
- It is not a representative snapshot: farms and shelterbelts were not randomly sampled, but chosen to have similar radiata pine age with a nearby native shelterbelt. Comparisons between natives and pine, and between the four farm types, are not matched for age or management.
- Only coarse woody debris, litter and herbaceous understorey were sampled for biomass. AGB was based on two DBH and height measurements per 100 m shelterbelt. BGB was based on root:shoot ratio and soil C on a 10 cm soil core. A C fraction of 0.5 was used to convert biomass to C.
- The Moore (2002) allometric equation was used for radiata pine, with no indication that topping was taken into account. Half of the radiata pine shelterbelts had been topped at 10 m or lower, and the usual approach would be to estimate the volume for a tree of full height and subtract the missing part. DBH was also taken at 1.3 m rather than the New Zealand standard of 1.4 m.
- Wood density variation with age is captured for radiata pine indirectly via the allometric equation, because height in forest trees is correlated with age. Since this is not the case for topped shelterbelts, the model would assume that a height of (for example) 5 m indicated a young tree and would therefore underestimate wood density. McKinley et al. (2000) showed that the trend of increasing wood density with age is found across a range of exotic plantation species used in New Zealand. However, all native shelterbelt species were assigned a constant wood density regardless of age.
- Some of the inputs appear unlikely (e.g. radiata pine age 25 with mean DBH > 150 cm).

The raw data could be reanalysed to provide a yield table for managed shelterbelts, but without destructive biomass sampling (or at least better representation of tree form by including top diameter) there will remain some doubt over the estimates.

There is an increasing amount of international literature on C stocks and stock changes in shelterbelts. Studies differ in terms of the pools included and the metrics presented (e.g. tonnes of dry matter, C or CO_2, and on a kilometre, hectare of shelterbelt or hectare of landscape basis). Most studies are based on physiological growth models or the application of allometric equations. Crossland (2015) reported biomass stock estimates for hedgerows as ranging from 18.3 $t{\cdot}CO_2{\cdot}ha^{-1}$ to 165 $t{\cdot}CO_2{\cdot}ha^{-1}$ (Falloon et al. 2004; Warner 2011; Robertson et al. 2012). Modelled results for above-ground biomass in unmanaged hawthorn and blackthorn hedgerows were 342 $t{\cdot}CO_2{\cdot}ha^{-1}$ and 480 $t{\cdot}CO_2{\cdot}ha^{-1}$, respectively,

[8] This information has been sought from the authors.

with an average sequestration rate of 6.5 to 22 $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}yr^{-1}$ (Crossland 2015). Pöpken (2011) estimated copses and hedgerows in Germany contained on average 158.3 $t{\cdot}CO_2{\cdot}ha^{-1}$.

d) Shelterbelt emission factors for New Zealand

In general, tree form in shelterbelts is not the same as in forests. Trees grown in the open are shorter, with greater taper and larger branches than trees grown in stands. Trimming will compensate for the greater branch biomass to some extent. Shelterbelts are also likely to be at a higher stocking than forests. Estimates of shelterbelt biomass made in New Zealand have used forest allometric equations and have not explicitly taken into account the effect of trimming and topping on tree form and biomass.

Dense hedges that are managed at a constant height and width will have limited sequestration potential once they reach the desired dimensions. The maximum stock may be similar for hedges of the same height and width, regardless of species. The time taken to reach that stock may also be similar, assuming that care is taken with establishment. Shelterbelts could then be maintained for decades with annual growth being pruned off.

The wide variety in shelterbelt design makes it difficult to assign appropriate emission factors. A possible approach is to define broad classes of shelterbelt, for example:

- i Non-woody (e.g. flax) or non-tree shrubs: apply GWB emission factor
- ii Trimmed, but not topped, tree species: apply ETS look-up tables
- iii Topped tree species by height classes; e.g.
  a) 1–2 m height (assume as for non-woody)
  b) 2–8 m height (assume a proportion; e.g. 1/3 of unmanaged shelterbelt c)
  >8 m (assume a proportion; e.g. 2/3 of unmanaged shelterbelt C).

Table D 5: Emission factors for shelterbelts in New Zealand

Vegetation type and area	Age at maturity/steady state (where appropriate)	Maximum stock at maturity/steady state $t{\cdot}CO_2{\cdot}ha^{-1}$	Stock assumed at time of clearance (if not max. stock)* $t{\cdot}CO_2{\cdot}ha^{-1}$	Time period assumed for sequestration	Mean sequestration rate ** $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}\ yr^{-1}$	Source(s)
Radiata pine – Gisborne*	28	807	807	28	28.8	ETS look-up table
Radiata pine – Canterbury / W. Coast**	28	515	515	28	18.4	ETS look-up table
Other softwoods	40	524	524	40	13.1	ETS look-up table
All hardwoods	20	526	526	20	26.3	ETS look-up table
Native regeneration	50+	323.4	184	50	6.5	ETS look-up table
GWB – permanent		222.09		28	7.9	NZ NIR***
	Average age	Average stock				
Natives	16	251.9			15.7	Welsch et al. 2016
Radiata	27	346.9			12.8	Welsch et al. 2016
Radiata/macrocarpa	Unknown	869				Czerepowicz et al. 2012

* Most productive ETS look-up table for radiata pine.

** Least productive ETS look-up table for radiata pine.

*** NIR – New Zealand's Greenhouse Gas Inventory 1990-2015. The National Inventory Report and Common Reporting Format. Ministry for the Environment, Wellington.

e) Shelterbelt Summary

Shelterbelts and hedges on farms serve different purposes and are managed in a variety of ways. Variation in species, height, width, number of rows, tree spacing, branch trimming and topping make it a difficult resource to characterise and model. There has been no destructive biomass sampling of shelterbelts in New Zealand: available estimates of C stocks in shelterbelts are based on the application of forest tree allometric equations that are unlikely to adequately represent shelterbelt trees, except possibly in unmanaged shelterbelts.

It is likely that shelterbelts rapidly sequester C from the time of establishment until they reach their desired final height. Unless left to attain their natural maximum height, repeated annual topping and trimming from this point on mean that the level of ongoing sequestration will be low. The average height and width of topped shelterbelts in New Zealand is unknown, as is the average useful lifespan for shelterbelts before they are replaced. A farm is likely to include both actively growing shelterbelts and shelterbelts that are more or less C neutral. Emissions from removed shelterbelts will also occur.

Reanalysis of the data reported by Welsch el al. (2016) is one option for improving estimates for topped and trimmed shelterbelts. The Scion 'timber belt' database has information for radiata pine, Japanese cedar and Leyland cypress, but for stem volume only in trees that have not been topped. Biomass sampling and/or improved sampling and measurement protocols of hedges covering a range of widths and heights would be required to more accurately determine emission factors.

As an interim estimate it is suggested that the estimates in Table D6 could be used.

Table D 6: Suggested values for maximum stock, age and sequestration rates in different types of shelterbelt

Shelterbelt type	Maximum stock t·CO2·ha^{-1}	Age	Sequestration rate t·CO2·ha^{-1}	Source
Non-woody or shrubs	222	28	7.9	NZ NIR
Untopped trees	515–807	28–40	13.1–28.8	ETS look-up tables
Low hedges	222	28	7.9	NZ NIR
Medium-tall hedges	364	28	13	Based on Welsch et al. 2016

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E. Small woodlots (Steve Wakelin)

a) Definition, characteristics and management

A typical New Zealand woodlot is a stand of trees, usually even-aged and managed for timber production under a clearfell harvesting system. The definition could be expanded to include stands of trees outside riparian zones and shelterbelts, with mixed ages and species (including natives) and with or without intended harvesting. Potentially this could include areas of naturally regenerated vegetation.

Under the ETS, woodlots qualify as forests if they meet the thresholds used for New Zealand's GHG inventory:

- i tree species capable of reaching 5 m height *in situ* (UNFCCC[9] minimum: 2−5 m)
- ii 30% canopy cover (UNFCCC minimum 10−30%)
- iii 30 m minimum width
- iv 1 ha minimum area (UNFCCC minimum 0.05–1 ha).

For this report we assume that the intent is to include woodlots that meet all of the forest criteria except the minimum size of 1 ha; for example, woodlots of 30 m width but less than 333 m length. If no minimum area is specified, then individual trees would be eligible. Even at the minimum UNFCCC scale of 0.05 ha, a single mature tree could achieve the required crown radius of 12.6 m (Pretzsch et al. 2015).

In the cases where the other forest criteria are not reached there are other approaches to take. For example, if the 30% canopy cover threshold is not reached, the 'woodlot' could be treated in the same way as space-planted poplars and willows used for soil conservation. This could also be applied to shade or specimen tree plantings. If the 5 m height threshold is not reached (e.g. non-forest species planted, or management intervention prevents the height being reached), the 'other vegetation' category could apply.

For this report we are concerned only with the establishment of post-1989 vegetation rather than the enhancement of stocks in vegetation that already existed in 1990 and are therefore not eligible in the ETS. Forest fragments of less than 1 ha are vulnerable to disturbance and may be a carbon source rather than a sink (e.g. Burns et al. 2011). Confirming stock changes in native forest remnants has proven to be difficult as the change is a small difference between two large stock estimates with their often large associated errors (e.g. sampling and measurement error).

The inclusion of carbon sequestration by small woodlots in the ETS also raises the question of deforestation liabilities for the removal of small stands of trees. Clearance of

[9] Decision 11/CP.7 Marrakesh Accord

vegetation areas of less than 1 ha is currently not considered deforestation as these are not considered forests. The credibility of the carbon market may require symmetry.

b) Classification for NZ UNFCCC reporting and accounting

Currently woodlots that meet the forest thresholds are classified as forests. Smaller woodlots are likely to be mapped with the surrounding land use if isolated (e.g. high-producing grassland) or may be mapped as 'grassland with woody biomass – transition' if there are other stands or scattered trees in the area.

c) Emission factor literature review

Allometric equations for forest tree species are specifically for trees grown in competition with neighbouring trees. These differ from open-grown trees, which are shorter, with larger diameters, wider crowns and heavier branching.

The effects of stand edge have long been recognised in the plantation forest industry because of its influence on log grade out-turn (Maclaren 2000). The difference in edge trees is visible as larger-diameter logs with larger branches, particularly where branches reach out into open space outside the planted forest area. The importance of the edge effect increases for smaller plots, where the edge affects a greater proportion of the stand area (Beers 1977). A number of factors are hypothesised to contribute to the edge effect, including resource availability in terms of light, water and nutrients, as well as trees' ability to take advantage of these resources at the stand edge and the effects of wind near open forest edges. Zavitovski (1981) showed that the edge effect in plantation poplar clones extended several rows into the stand.

As well as the effect of water and nutrient availability, the effect of wind loading near the stand boundary is more pronounced due to the edge effect (Somerville 1981). The increased stem and root growth of wind-loaded trees has been shown to affect root biomass growth for two common northern hemisphere conifer species (Stokes 1994). Emission factors developed for forest trees will be less suited to trees in shelterbelts and small stands with a higher proportion of edge trees. There is some research and data comparing edge and non-edge trees that could be used to adjust the prediction of biomass components with forest-type models (e.g. Berg 1973).

Short rotation coppice (SRC) crops grown for bioenergy are another potential sub-category, although it may not be economical to grow these in areas less than 1 ha. Harvesting of SRC willow in New Zealand has been recommended to take place every 3 years over an 18-year cycle (Snowdon et al. 2013), so that the only build-up of biomass carbon is in the roots. Hauk et al (2013) reviewed 37 mainly European SRC trials, and Krasuska and Ronsenqvist (2012) assumed a 25% lower yield level when translating research trial results into commercialised cultivation. Taken together these results suggest a mean total biomass stock of about 23 $t \cdot CO_2 \cdot ha^{-1}$. These crops would meet the forest thresholds but sequester little carbon per hectare. Crops such as *Miscanthus* are harvested annually so there is no ongoing above-ground C sequestration once the full crop area has been established.

Carbon sequestration by individual specimen trees, particularly in urban areas, is a popular field of research overseas (e.g. Russo et al. 2014), but there has been little biomass sampling of trees outside forests in New Zealand. Schwendenmann and Mitchell (2014) investigated carbon storage by native trees in an urban park in Auckland, but the trees were effectively a densely stocked 27-year-old regenerating native forest.

d) Proposed woodlot emission factors for New Zealand

The simplest option would be to relax the minimum area thresholds for meeting the ETS forest definition. Then the ETS look-up tables could be used to estimate carbon uptake and emissions from the establishment and removal of small woodlots, exactly as for large woodlots. Species would map to the existing forest options of radiata pine (by region), Douglas fir, other softwoods, exotic hardwoods and natives. Stands of shrubs that do not meet the height threshold would be excluded, as would widely spaced trees that do not meet the canopy cover threshold.

The ETS look-up tables provide sequestration rates that vary by age and allow for variable rotation lengths and delayed emissions from harvest residues if the stand is replanted. Mean sequestration rate over typical rotations is shown in Table E1 for the most and least productive radiata pine region and the other species. These values are given as a comparison with the other vegetation types considered: in practice, the look-up table would be used rather than an EF.

Table E 1: Emission factors for small woodlots based on ETS look-up tables

Vegetation type and Region	Age at maturity / steady state (where appropriate)	Maximum stock at maturity/ steady state $t{\cdot}CO_2{\cdot}ha^{-1}$	Stock assumed at time of clearance (if not max stock) $t{\cdot}CO_2{\cdot}ha^{-1}$	Time period assumed for sequestration	Mean sequestration rate $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}yr^{-1}$	Source(s)
Radiata pine – Gisborne*	28	807	807	28	28.8	ETS look-up table
Radiata pine – Canterbury/ W. Coast**	28	515	515	28	18.4	ETS look-up table
Douglas fir	40	751	751	40	18.8	ETS look-up table
Other softwoods	40	524	524	40	13.1	ETS look-up table
All hardwoods	20	526	526	20	26.3	ETS look-up table
Native regeneration	50+	323.4	184[a]	50	6.5	ETS look-up table

* Most productive regions for radiata according to ETS lookup tables

** Least productive regions for radiata according to ETS lookup tables

Sequestration rates actually achieved could vary markedly. In general the ETS look-up tables for plantation species are conservative and are based on the genetic stock and lower stocking rates used in the past. Much higher C stocks can be achieved in woodlots, particularly when managed with carbon sequestration. Whiteside et al. (1997) suggested optimal final crop stockings of 250 stems ha^{-1} for pruned regimes and 350 stems ha^{-1} for unpruned regimes. Average final crop stocking has increased steadily since then, in part because the proportion of stands receiving pruning has declined. Watt et al (2016) reported that surveyed forest owners aimed for an average stocking for unpruned regimes of 505 stems ha^{-1}, while modelling showed that the average optimal stocking to produce high value small-branched logs was 603 stems ha^{-1}. Modelled regimes at these high stocking rates predict two to three times the sequestration rates indicated in the lookup tables.

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F. Revegetation of retired land not otherwise eligible for ETS (Larry Burrows & Steve Wakelin)

a) Definition, characteristics, background and management

Most of New Zealand was once covered by forests (McGlone 1989), so there is potential for much of the country to revert to trees. Although there is significant potential for restoring indigenous forests (Dymond et al. 2013; Carswell et al. 2015), reversibility to original states is very slow or limited in some circumstances (Walker et al. 2009a). Some areas are unlikely to revert to forest (under the ETS definition of forest) in the foreseeable future due to environmental limiting factors (MfE 2012, Figure 1), or management actions may deter forest from developing.

Environmental limits that will slow tree establishment and growth include locations that are too hot, too cold, too wet, too dry, too infertile, lack seed, are too exposed, salty, windy, high elevation, or have too much competition from other vegetation. In these cases vegetation will struggle to meet ETS compliance rules of >30% cover of forest canopy over 5 m and >30 m in width in any hectare in any sensible time period.[10] This includes lands often found on farms in minor amounts, such as alpine or subalpine grasslands and shrublands; frost flats; thermal areas; marshes, mires or wetlands; gumlands and pakihi; ultramafic low-fertility lands; saline, coastal areas or estuaries; sand dunes; lake, river or ephemeral turfs; braided riverbeds; or other land exposed to frequent disturbance, such as screes on steep slopes (Singers & Rogers 2014). All the land-cover classes discussed elsewhere in this report also fit within this class.

Land that is maintained as non-forest because it is limited by management actions may include: where tree species are topped or felled before reaching 5 m; where land is managed for shrubland crops, such as mānuka or flowers, and transition to forest status is prevented; nurseries; areas that are regularly burned or sprayed; and areas with intensive grazing by domestic or pest animals. Shrubland that is still subject to grazing is assumed to be periodically cleared to promote pasture growth. This means that the land is not considered to have the potential to reach forest status and the clearance is not regarded as deforestation.

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[10] https://www.mpi.govt.nz/growing-and-harvesting/forestry/forestry-in-the-emissions-trading-scheme/

Figure F 1: Areas of New Zealand where environmental factors (e.g. elevation, wind, temperature, soil, moisture, exposure) are considered to limit growth to forest stature in 30–40 years are shaded dark green (from MfE 2012). Areas on farmland that are not eligible for registration into ETS include far more land than this (e.g. grasslands, small lots, shelterbelts, riparian restoration, and areas actively deterred from reversion).

Land not eligible for ETS is based on current ETS / Permanent Forest Sink Initiative eligibility rules for carbon trading schemes that allow landowners to account for net sequestration by forest that began after 1990. Hence land will *not* be eligible for the ETS if it comprises areas that were already forested prior to 1990, retired areas of trees less than 1 ha or less than 30 m width, or areas of trees that won't achieve 30% cover on any hectare.

The exception to these conditions is retired lands without trees. Common vegetation land cover in retired areas without trees includes grasslands (dominated by either short fescue/poa or tall *Chionochloa* snow tussocks) and short shrublands, or commonly, a mixture (sometimes referred to as 'Grassland with Woody Biomass'). Retired lands that fit this class are extensive. For example, through the tenure review process the grazing leasehold of c. 330,000 ha has been retired and relinquished to the Crown in the past 25 years.

The question that then arises is what are the EFs associated with land retirement (e.g. removal of grazing) on lands that are unlikely to revert to forest?

Other non-ETS revegetation on farmland includes areas where forest establishment is possible but can be very slow, along with the vegetation classes that are the subject of this report. MPI excludes land from the ETS that will take 30−40 years to revert to forest (MfE 2012). Slow tree establishment may result from: environmental conditions (e.g. hot, cold, wet, dry) limiting growth (Walker et al. 2009a); herbivory by livestock (Buxton et al. 2008) or wild pests (Holdaway et al. 2012); or few or no seed sources from tall trees within the range of a few kilometres (Mason et al. 2013b). Less commonly, filtering/competition by smothering lianas or tree ferns may also slow forest succession (Richardson et al. 2014; Brock et al. 2018).

b) Emission factor literature review

Land that was already in forest by 1990

Many landowners look at the carbon stock contained as forested land on their property and consider whether there is any carbon sequestration occurring and whether they could benefit from it. The short answer is generally nil and no, because although there may be large stocks in mature forest, but it is treated as being in a net stable state of near-zero sequestration. This is because, on average at large scales, new growth by existing trees in an old-growth forest is approximately matched by mortality of old stems, fallen branches and deadwood (woody debris and litter) decomposition (Shugart 1984). In a given situation this balance might actually be slightly positive or slightly negative depending on many variables such as the age of the forest, the impacts of pests and diseases, climate differences from year to year (including concentrations of greenhouse gases themselves), changing species composition and different wood density, and environmental impacts such as storms or droughts.

To tease out this balance and attempt to determine whether there is any very small change in a very large biomass carbon stock (e.g. on New Zealand indigenous forests as a whole, covering c. 8.2 million ha) is very difficult and has been the subject of considerable research effort over the past couple of decades (Coomes et al. 2002; Coomes et al. 2012). Parallel programmes track carbon stocks in New Zealand's plantation forests and soils, two other very large carbon stocks that change only slowly.[11] To determine whether there is a small change in a large stock for stands of forest trees on individual properties would be driven by cost:benefit limitations to the owner.

Other lands

Data on sequestration rates relating to this wide range of ecosystems and vegetation classes is very limited in New Zealand. By definition, the carbon stock potential on lands that are incapable of attaining forest status is low. Currently the greenhouse gas inventory recognises two classes of shrubland or 'Grassland with Woody Biomass': one for transitional vegetation in managed landscapes, and one for permanent shrubland. Emissions factors have been estimated from the available data but are not based on

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[11] http://www.mfe.govt.nz/publications/rma/measuring-carbon-emissions-land-use-change-and-forestry-new-zealand-land-use-and.

nationally representative sampling (Wakelin & Beets 2014) and may not be applicable to the vegetation types in this category on farmland. Assessing carbon stocks in shrublands worldwide is difficult because of their variability, unclear margins (crossing over into both grassland and forest) and often dynamic nature.

c) Current classification for NZ UNFCCC reporting and accounting

Multiple vegetation classes that are included in other broad classes include the high- and low-producing grassland categories ‘Grassland with Woody Biomass’ or ‘Other Land’. Emissions factors for ‘Other Land’ are not estimated and are treated as zero by MfE (2017).

d) Proposed EFs for New Zealand and data

The vegetation and land classes that may be retired and not eligible for ETS are hugely varied. The consistent feature of almost all these classes is their low productivity and C sequestration potential. There is also a lack of information on their sequestration rates, apart from alpine snow tussock grassland, for which there are quite a few study results for productivity over a long period.

For these reasons we see little value in proposing anything other than the current New Zealand national accounting values (MfE 2017). A weighted average EF for the broad category cannot be calculated without knowing the proportions of the various vegetation types and their C stock potential.

Table F 1: Emission factors for retired land based on MfE (2017)

Vegetation type	Max. stock at maturity/steady state $t{\cdot}CO_2{\cdot}ha^{-1}$	Stock assumed at time of clearance (if not max. stock)* $t{\cdot}CO_2{\cdot}ha^{-1}$	Time period assumed for sequestration (yrs)	Mean sequestration rate ** $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}yr^{-1}$	Source(s)
Non-ETS compliant	NA	NA		0	
Alpine shrubland	60.57		>50		GHG NIR* GWB** – permanent
Tussock	NA				
Frost flats; thermal areas	13.05		>50	0.47	GHG NIR – GWB transitional
Disturbed	-		-	-	
Low fert.	13.05		>50	0.47	GHG NIR – GWB transitional
Management intervention	60.57				GHG NIR GWB – permanent

*GHG NIR – MfE 2017. New Zealand's Greenhouse Gas Inventory 1990–2015. The National Inventory Report. Wellington, Ministry for the Environment.

**GWB – Grassland with woody biomass. Wakelin & Beets (2014)

e) Retired land Summary

In some cases farmland that is retired from grazing will eventually revert to native forest, as long as a seed source is available, so is then likely to qualify as forest land. Retired land that does not meet the ETS requirements includes a wide range of vegetation classes and ecosystems. The reasons that land may not be eligible for the ETS may be a function of environmental limitations or land management practices. Large areas of New Zealand that were cleared of forest or trees may be very slow to recover.

Environmental controls relating to elevation, temperature, moisture, exposure and soil fertility also affect retired land that does not meet the ETS forest definition. In these situations C sequestration will probably be negligible or small due to the same conditions that limit reversion to forest. Improving revegetation and vegetation growth rates of retired land that otherwise would not make a forest will be prohibitively expensive due to the tending that would be required.

To develop suitable EFs for these vegetation classes will be a major undertaking due to the wide range of ecosystems and the likely low growth rates.

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G. Overall Sequestration Potentials

a) National and Regional Potentials

For some of the vegetation classes it is instructive to carry out preliminary conceptual models based on broad national estimates of area and median assumptions of C sequestration to get an idea of the size of the 'potential prize'. There will be a wide range of abilities by different farm types and locations to adopt particular potentials on individual properties, so we have not divided them to a finer scale. For example, not all farms have streams or wetlands, some farms are flat so have no need for erosion control pole planting, large hill farms will find it easier to allocate land for woodlot planting.

It should be kept in mind that the potential net carbon benefit is a combination of area of vegetation multiplied by sequestration rate (for a given period until maturity when it approaches zero), MINUS the C stock of cleared vegetation, all after some fixed point-of-time baseline. So the period since any baseline and changes during that time will be important. There is no obvious baseline date for any of these classes or basis of confirming their areal extent before now. Sequestration rates and C stocks are uncertain as discussed in previous sections and the annual areas of vegetation or wetland gain and loss even more so.

These potentials will only be achieved as genuine biomass sequestration offset sites by additional new land allocated to them. Typically 0.4 – 2.5% of New Zealand farmland in the cases described. For such a change in land-use of any farmland would require a favourable cost/benefit economic analysis that is not within the remit of this report.

Wetlands

A national wetland area of 2.5% could be seen as a trade-off target between productive farm area reduction and water quality improvement (e.g. Tanner et al. 2013). About 250,000 ha of wetland remain across New Zealand. A subset of those occurs on farmlands and a proportion of those will be peat mires that may sequester C. If the extent of wetlands on farmland was 25% of the total and 50% of those were peat mires, and that area was multiplied by the 2.0 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$ mid-range net CO_2eq sequestration rate from the Goodrich et al. (2017) Kopuatai peat bog study (range 0.78–3.55 $t.CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$), then an estimate of 0.063 $Mt{\cdot}CO_2e{\cdot}yr^{-1}$ could be sequestered by New Zealand's on-farm wetlands. Clearly there are a number of uncertainties in this estimate, but it may be useful for considering whether it is worthwhile to invest further to undertake research to fill information gaps to increase the confidence in the estimate.

Taranaki Ring Plain riparian example - Taranaki Ring Plain and coastal terraces (Elizabeth Graham)

One of the only New Zealand Regional case studies on the biomass C potential of Riparian establishment is based on data from Taranaki Regional Council (TRC).

TRC has a long-term focus on riparian fencing and planting along streams in the dairy-dominated area of the Taranaki Ring Plain and coastal terraces (2,803 km^2, pers. comm., Don Shearman, TRC). TRC's Geographic Information System (GIS) database on riparian fencing and planting documents, from 2000 onwards, the planting dates and widths within riparian fenced areas of four vegetation classes: native and exotic low-growing vegetation, and native and exotic trees and shrubs. We used these data for the 1124 km of streams planted during 2000–2017 with native (93%) and exotic hardwood (7%) trees and shrubs that have the potential to grow to over 5 m high forests (pers. comm., Don Shearman, TRC), in order to estimate C sequestration that has been and could be achieved by riparian planting.

The average fenced width from the stream edge of these riparian areas was 11.4 m on each side, of which an average 3.2 m was established with forest vegetation (after adjusting for the presence of low-growing vegetation in some plantings by reducing the actual forest species planted area by 1/3 and reducing the width by 0.5 m to account for animal browsing along the fenceline; further details are in Appendix G1). These data were combined with the CO_2 sequestration rates, by age, for exotic hardwoods and indigenous forest in MPI (2017) Look-up tables to estimate total riparian CO_2 sequestration for new riparian areas planted with trees and shrubs between 2000 and 2017 (see Appendix G1 for calculation details).

We estimated that 57 tonnes of $CO_2 \cdot ha^{-1}$, or 20,624 tonnes CO_2 total, were sequestered by the riparian vegetation planted between 2000 and 2017 (Figure G1). This indicates an average sequestration per year of approximately 3.4 tonnes $CO_2 \cdot ha^{-1}$.

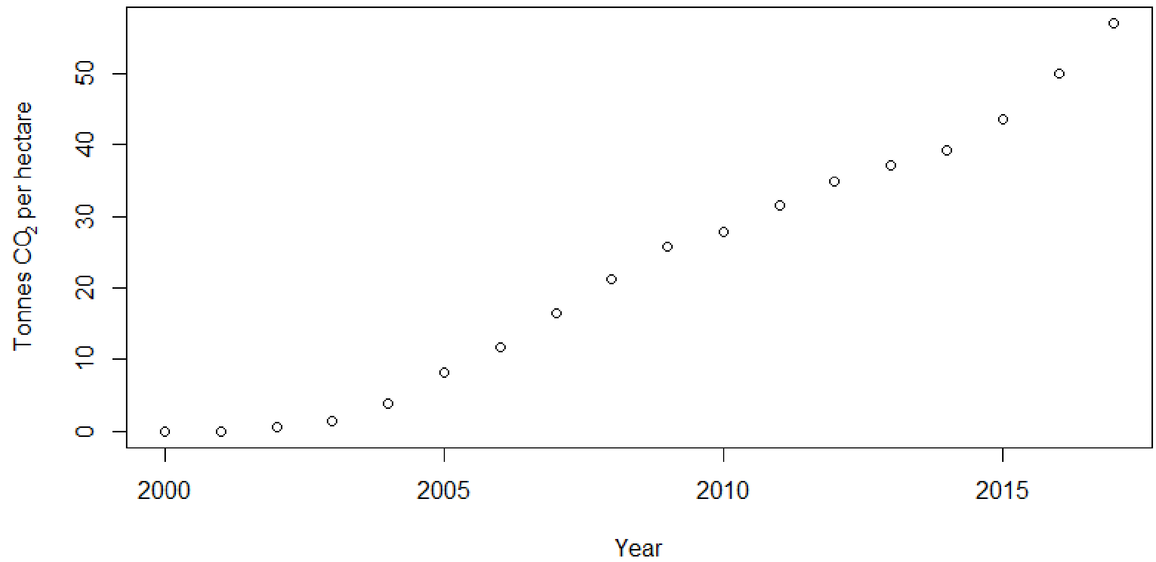

Figure G 1: Estimated cumulative annual CO_2 sequestration per hectare in riparian forest along the Taranaki Ring Plain and coastal terrace streams planted from 2000 to 2017, based on Taranaki Regional Council data.

The potential for CO_2 sequestration by riparian forest plantings was explored further by using the MPI (2017) look-up tables and the TRC planting rates and native/exotic mix from 2000 to 2017, and setting riparian forest planted widths of 1 to 10 m on each side of the streams (Figure G2). The estimates for 10 m wide buffers of planted forest (Figure G2) indicate there is potential for substantially higher CO_2 sequestration if almost all of the

11 m-wide fenced area of the TRC-planted buffer sites were planted with forest species (c.f. the average of 4 m planted with a mix of low-growing and forest plants in Figure G1).

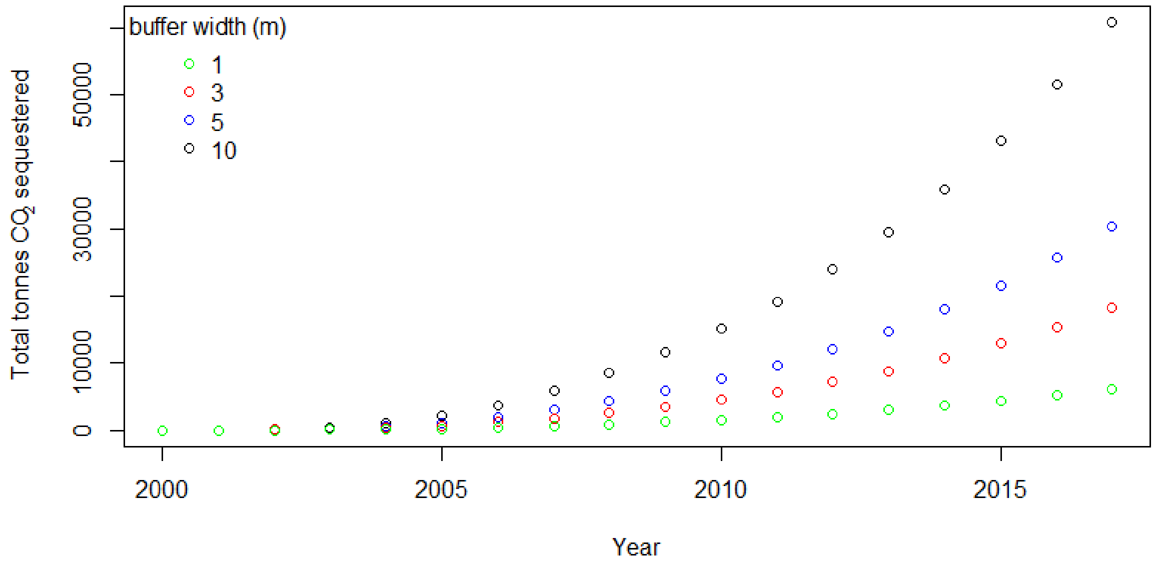

Figure G 2: Simulated total CO_2 sequestration within riparian areas along 1,124 km of Taranaki streams for planted forest widths of 1 to 10 m.

Note that Taranaki has a relatively high stream density (e.g. 3.2 km/km^2 in the Waiokura (Wilcock et al. 2009) compared with a national average of 1.6 km/km^2 (= 425,000 km/268,021 km^2), resulting in a relatively high potential for riparian areas to sequester CO_2 per hectare of farm area.

Riparian potential

About half of New Zealand's 425,000 km of rivers and streams are found in agricultural (43%), forestry (5%) and urban (1%) landscapes (http://www.doc.govt.nz/about-us/statutory-and-advisory-bodies/nz-conservation-authority/publications/protecting-new-zealands-rivers/02-state-of-our-rivers/overview-of-new-zealand-rivers/). If 50% of those were established in 10 m wide riparian buffers on either side of the river or stream that would amount to 212,500 ha nationally (c. 1.4% of all farmland) that could, according to a sequestration rate used in the above scenarios from Taranaki Ring Plain, potentially sequester c. 3.4 $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}yr^{-1}$= 0.72 $MtCO_2e{\cdot}yr^{-1}$. This value must be treated as indicative only as it is based on a number of unproven assumptions. Ten metres was used as an indicative width as it is required under National Environmental Standard for Plantation Forestry for streams > 3m wide, is indicated as the setback distance for average riparian fences in Taranaki (even though just c. 3 m is planted), will improve the buffer of soluble below-ground contaminants and wider riparian strips are being seen in new farm planting plans (see Case Study 3, Fig H3). There are other obvious limitations to enacting such an outcome. Sequestration will be strongly affected by species composition and whether they are grasses, sedges, shrubs or trees. Is establishment of riparian strips on 50% of stream banks realistic? This rate is not real but it indicates there may be some potential for sequestration by riparian strips of those dimensions on farms with streams.

Shelterbelts and Small Woodlots

Welsch et al. (2016) estimated from a survey of farmer landowners in Canterbury that there was support for increasing planted areas of trees by most farmers. They indicated that an increase of 4 % of land area into planted riparian strips and shelterbelts was supported according to landowner preferences, although this varied among farmer groups.

Small Woodlots at a national scale also offer considerable potential as a C offset for farmland. In order to achieve 1.0 Mt CO_2 yr^{-1} sequestration, until they mature about 2050, a little over 4 ha of every 1000 ha. (0.4 % of agricultural land) would need to be newly established in a mix of exotic and/or native woodlots (i.e. based on 16 $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}yr^{-1}$ median).

Estimates of the potential area of extensive erodible hill country grazed land that could be available for afforestation range from 0.7 million to 2.9 million ha (Watt et al 2011, Mason and Morgenroth 2017). This amounts to c. 5 – 20% of total agricultural land area that could be considered for wide-spaced pole planting.

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Appendix G1: Taranaki Ring Plain example riparian carbon sequestration calculation procedure

1 Calculate the lengths and widths of riparian plantings extracted from TRC's GIS database, along with the year each planting was completed and the type of vegetation (i.e. trees, shrubs or low-growing sedges, etc., and native or exotic).

2 Estimate the width of planting by subtracting distance from stream to vegetation from distance from stream to fence. Subtract an additional 0.5 m to account for grazing by livestock reaching inside the fence.

3 As part of this process, eliminate fences with no corresponding vegetation and vegetation with no corresponding fences (approximately 5% on each bank).

4 If low vegetation is also present in the same spot, decrease widths by 1/3 to reflect the space taken up by those plants (as per Don Shearman's recommendation).

5 Multiply length by width and convert area to hectares.

6 Calculate age of each vegetated patch as year planted subtracted from 2017.

7 Multiply areas by values from post-1989 forest land MPI look-up tables for tonnes of CO_2 sequestered per hectare for indigenous forest or exotic hardwood (values vary with vegetation type and age).

8 Sum all segments to get total tonnes of CO_2 sequestered.

9 Divide by total hectares to get tonnes per hectare.

H. Farm Case Studies (Alec Mackay)

a) Introduction to Farm Case Studies

Farm case studies illustrate the opportunities and challenges of accounting for carbon from on-farm vegetation using C sequestration rates sourced from the literature and collated here as part of the current project. From discussions with the working party three case studies were decided on: one for an intensive sheep and beef operation in North Island hill country; one for an intensive lowland sheep and beef operation in Southland; and the third a dairy farm operation in the Waikato. The farm case studies are based on vegetation data sets collected as part of existing on-farm planning and reporting processes and management programmes in place to address existing industry challenges that include soil erosion, water quality, protection of indigenous biodiversity and animal welfare.

b) Background

Farm plans are increasingly being used as a vehicle for affecting change/ implementation of policy. Some farm planning processes already collect a large amount of information on the history, areal extent, type and condition of existing vegetation, and include information on future planting enhancement programmes. This approach could be used with some additional measures for collecting the required data for quantifying existing C stocks on-farm, the basis for estimating C sequestration rates and creating a C budget for the farm.

Farm planning underpinned by land evaluation has its origins in soil and water conservation in New Zealand. The approach has received extensive and successful application in New Zealand and overseas since at least the 1940s (Manderson & Palmer 2006). Over the decades land evaluation and planning has expanded to address on-farm issues beyond soil erosion, to include, for example, nutrient management, riparian management including fencing and planting, and water quality. In doing so has become an increasingly important part of farm business planning.

When applied according to original principles, farm planning is also useful for helping landholders identify and evaluate how changes in land use and management can result in profit and production gains and environmental enhancements. This is reflected in the whole farm plans of Horizons Regional Council, which forms part of the Sustainable Land Use Initiative, and includes an analysis of the farms business in addition to an assessment of the biophysical resources of the farm (Manderson et al. 2013). The Land Environment Planning Tool Kit of Beef and Lamb NZ also addresses production as well as environmental issues on-farm in the Level 2 and 3 farm plans (Synge et al. 2013). The DairyNZ riparian planner enables dairy farmers throughout the country to create a riparian management plan to fence, plant, and protect waterways with our new online tool https://www.dairynz.co.nz/environment/waterways/riparian-planner/

Regional land and water resource management plans increasingly use farm plans as vehicles for implementation. In some regions, including Hawke's Bay (https://www.hbrc.govt.nz/assets/Document-Library/Tukituki/ Tukituki-Plan-Change-6.pdf) and Canterbury (http://www.canterburywater.farm /zones/), a farm plan is required as part of the regulatory framework for addressing impacts on water bodies. They are also being currently developed to include indigenous biodiversity and capture services beyond food and fibre (Mackay et al. 2018).

The case studies below draw on vegetation data collected on-farm as part of existing planning and reporting processes.

c) Case Study 1. Hill country sheep, beef and deer operation (Southern North Island)

Location. Upper Pohangina Valley, in close proximity to the Ruahine ranges with an elevation from 500 m to 600 m.

Size: 412 ha. The farm has a range of landscape units (LUC Classes 2 to 8) typical of large tracts of Southern North Island hill country in sheep and beef production. Annual rainfall is 1200 mm/year. Winters can be cold and extended.

Operation: Intensive sheep breeding and finishing, with beef cows and hinds for venison production. 885 ewes, 50 rams, 275 hoggets, 60 beef cows, 29 heifers, 320 hinds.

Greenhouse gas emissions (methane, nitrous oxide, carbon dioxide) from this farming operation would be at the upper range of emissions for sheep and beef operations (2–4.5 $t{\cdot}CO_2\ e{\cdot}ha^{-1}{\cdot}yr^{-1}$) reported by Smeaton et al. (2011) and Mackay et al. (2011).

Vegetation: The farm has 270 ha in pastures. In excess of 1,000 poplar and willow poles have been planted on LUC Class 6 and 7 land over the last 10 years. The balance is in established native and regenerating bush which is under a QE II covenant. There is a small plantation of Douglas fir *(Pseudotsuga menziesii)*.

Table H 1: Age and area of regenerating bush, wide space-planted poplar and Douglas fir woodlots, and the amount of carbon sequestered, annually and in total, per hectare and as a percentage of the GHG emissions from the livestock on the farm under two carbon sequestration scenarios

Vegetation	Age (yr)	Area (ha)	C sequestration rates $t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}yr^{-1}$			
			per ha**	total	per ha***	total
Regenerating bush	>50	124	0.0	0.0	0.0	0.0
Space-planted (poplar)*	0-10	70	0.25	17.5	1	70
Woodlot (Douglas fir)	3-5	6.4	6.5	41.6	26.3	168.3
Total C sequestered ($t{\cdot}CO_2{\cdot}yr^{-1}$)				**59.1**		**238.3**
Total C emitted from farming operations ($t{\cdot}CO_2{\cdot}yr^{-1}$) ****		270	4.5	1215	4.5	1215
Fraction of GHG emissions emitted from farming operations				4.9%		19.6%

*Over approximately 70 ha, 1,000 poles have been planted over the last 10 years.

**Assumes low rates of C sequestration/ha for the regenerating bush, space-planted poplar and woodlot.

***Assumes medium rates of C sequestration/ha for the regenerating bush, space-planted poplar and woodlot.

****Assuming emissions from the livestock operation are at the upper end for a sheep and beef operations (4.5 $t{\cdot}CO_2e{\cdot}yr^{-1}$)

Of the two scenarios that were explored, it was assumed in the first that the regenerating bush was sequestering negligible C, because of its age and for the reasons described in section F(c). Both the spaced plantings and woodlot plantings are still young, which is the reason for assuming low sequestration rates for these two plantings in the first scenario.

The higher C sequestration rates for the spaced plantings and Douglas fir in the second scenario recognises that as these plantings age sequestration rates will increase. Under the second scenario the total amount of C sequestered would offset c. 20% of the annual GHG emissions generated by the livestock operation on the farm. The small woodlot offers the single biggest contribution to the budget.

Critically it must be remembered that in common with all biomass offset schemes that as the wide space planted trees and/or woodlots approach maturity the C sequestration rate drops towards zero, which would require new planting of additional land to maintain a C sink. In the case of harvested small woodlots it would result in a C loss, i.e. it would add to farm emissions when harvested and for the next ten years as the harvest residues decay, even if replanted. It can be thought about as sequestering about 16 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$on average over 25 years, which is about when the long term average C stock is reached. Then it would fluctuate around that equilibrium level in perpetuity providing no new net sequestration.

The influence of vegetation type on soil carbon stock change was not considered as part of the budget in the current study.

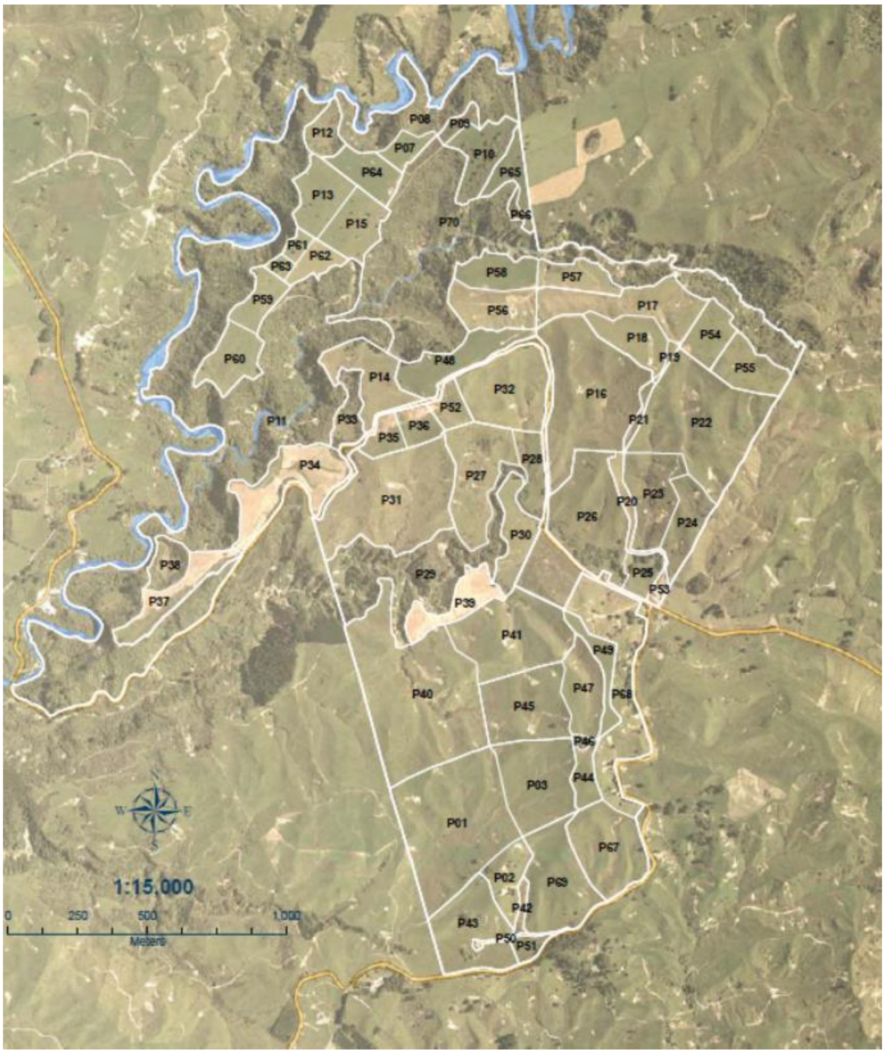

Figure H 1: Paddock map of the Pohangina Farm showing the topography and main vegetation types with large areas of native forest along the main water courses.

d) Case Study 2. Intensive lowland sheep operation (Southland)

Woodlands Farm (AgResearch)

Location: Woodlands farm is located 15 k east of Invercargill. It is on the intensively farmed southland plains and, because of its location and characteristics, is typical of the highly developed farms areas of the region involved in intensive sheep production.

Size: 230 ha (effective area 225.4 ha). Flat to gently undulating, with predominant soils Brown, with gleyed-soils adjacent to streams. Average rainfall 1050 mm evenly spread throughout the year

Operation: Intensive sheep breeding and finishing system Wintering 2000 ewes, 130 rams, 750 hoggets, 50 deer.

Greenhouse gas emissions (methane, nitrous oxide, carbon dioxide) for an operation of this type emissions would again be above the upper range in emissions for sheep and beef operations (2–4.5 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$) reported by Smeaton et al. (2011) and Mackay et al. (2011). The estimate for this operation was 5.125 $tCO_2\ e{\cdot}ha^{-1}{\cdot}yr^{-1}$.

Vegetation: The farm pasture is sown in with improved forage germplasm (predominately ryegrass and white clover). The farm has extensive shelter belt plantings that total 15.3 km, along with a small area of native bush and some riparian strips.

Shelter belts in the form of hedge rows include Pines (*Pinus radiata*), Blue gum (*Eucalyptus* sp.), Macrocarpa, Leyland cypress (Cupressus × *leylandii)*, Poplar (*Populus* sp.), and native trees. There is also a riparian area but this is primarily in pasture.

All the shelter belts are pruned on a regular basis to retain a defined hedge row form to limit spread and height. Most are more than 40 years old. The exception would be the *Leylandii*, which are 15–30 years old.

Because of the age of the trees and the fact the hedge rows are pruned on a regular basis, the amount of C sequestered each year is likely to be nil or very small.

Table H 2: List of the tree species, age length, width and area of each shelterbelt, and the amount of carbon sequestered as a percentage of the GHG emissions from the livestock on the farm under three carbon sequestration scenarios

Vegetation	**Age (yr)**	**Length (m)**	**Width (m)**	**Area (ha)**	**C sequestration rates ($t{\cdot}CO_2{\cdot}ha^{-1}{\cdot}yr^{-1}$)**		
					0*	1**	6.5***
Pines: *Pinus radiata*	>40	3,456	3	1.037	0	1.037	6.739
Blue gum: *Eucalyptus*	>40	3,912	3	1.174	0	1.174	7.628
Macrocarpa	>40	526	3	0.158	0	0.158	1.026
Leylandii	15−30	3,140	3	0.942	0.942	0.942	6.123
Poplar: *Populus*	>40	2,772	3	0.832	0	0.832	5.405
Native trees	>30	1,491	3	0.447	0.4473	0.447	2.907
Total		**15,297**		**4.589**	**1.389**	**4.589**	**29.829**
Emissions per ha from farming operations					5.125	5.125	5.125
Total emissions from farming operations				225.4	1155	1155	1155
Fraction of GHG emissions emitted by livestock					0.12%	0.39%	2.53%

* Assumes pines, blue gum, macrocarpa and poplar are not sequestering C, and Leylandii and native trees are sequestering 1 t·CO2·ha−1·yr−1.

** Assumes all shelterbelts are sequestering 1 t·CO2·ha−1·yr−1.

*** Assumes all shelterbelts are sequestering 6.5 t·CO2·ha−1·yr−1.

Three scenarios were explored. In the first it was assumed that the pine, blue gum, macrocarpa and poplar hedge rows were not sequestering C because of their age and also because of the fact they are pruned on a regular basis, while the Leyland cypress hedges (age) and native trees plantings (slow growing) were still accumulating a small amount of C (1 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$). Under this scenario the total amount of C sequestered amounts to an offset of 0.12% of the annual GHG emissions generated by the livestock operation on the farm. In scenario 2 and 3 the C sequestration rates were assumed to be 1 and 6.5 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$, respectively for all shelterbelt plantings. Even at a C sequestration rate of 6.5 $t{\cdot}CO_2e{\cdot}ha^{-1}{\cdot}yr^{-1}$, the total amount of C sequestered amounts to an offset of 2.53% of the annual GHG emissions generated by the livestock operation on the farm. This farm provides a good example of a case where there is a large amount of standing biomass carbon, but nil or little additional C is being sequestered.

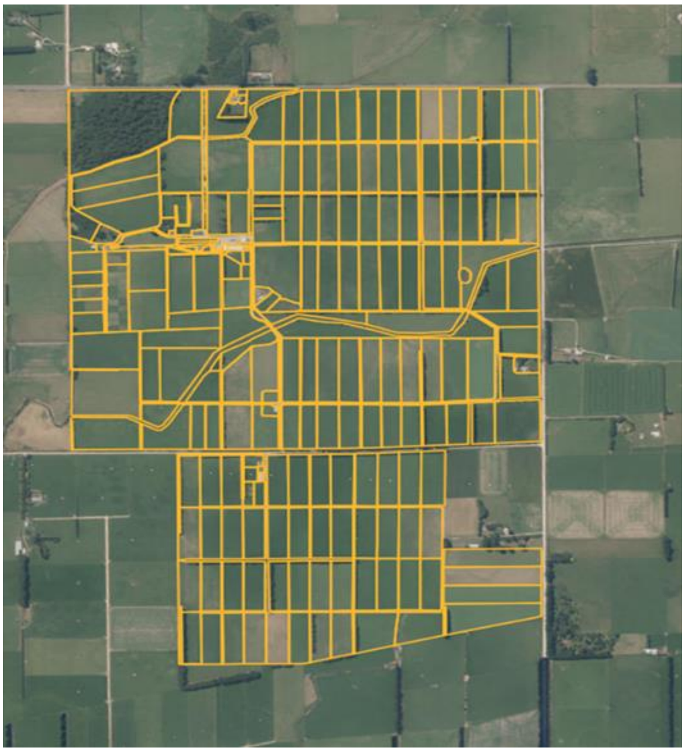

Figure H 2: Paddock map of the Woodlands Farm showing layout of the property and some examples of shelterbelts that are a feature of this landscape.

e) Case Study 3. Dairy operation (Waikato)

Location: Waikato close to Hamilton

DairyNZ Scott Farm

Farm profile: The home to research on effective farming systems for dairy farmers, with particular relevance for pasture-based Waikato farming. DairyNZ's larger scale farming systems research trials are based at Scott Farm. Current projects include Forage Value Index (FVI) and Improved Pasture Performance and Forages for Reduced Nitrate Leaching.

Size of the milking platform: Total area 138.95 ha, with effective area 116.22 ha. Flat to easy rolling with soils including Peaty loams, sandy loams, and silt loams. Annual rainfall is 1250 mm.

Operation: 400 dairy cows (subject to scientific requirements). 3.1 cows/ha, milk production of 931 kg Milk Solids/ha. Seasonal supply.

Greenhouse gas emissions: The amounts of methane, nitrous oxide and carbon dioxide produced by dairy cows are highly correlated with milk production (Smeaton et al. 2011, Fig. 6). They reported GHG ranging from 6.0 to 11.0 t·CO_2e·ha^{-1}·yr^{-1}. The current operation at Scott farm is assumed to be in the middle of this range ((8.5 t kg·CO_2e·ha^{-1}·yr^{-1}).

Vegetation: The farm is sown in modern forage germplasm. There is a small area of native bush on the farm and a plan in place to plant the riparian margins of the stream network of the farm.

The proposed riparian plan includes grass filter, lower bank planting (native grasses mostly, e.g. *Carex* spp., *Juncus* spp at 1-m spacing) and upper bank planting (native woodier shrubs and trees mostly, e.g. *Coprosma* spp., *Pittosporum* spp., *Leptospermum* spp. at 1.5-m spacing).

Table H 3: Length and area of the proposed riparian programme and the amount of carbon that could, in the future, be sequestered annually as a percentage of the GHG emissions from the livestock on the farm under three carbon sequestration rate scenarios

Vegetation	**Length (m)**	**Area (ha)**	**Emissions (t·CO_2e·h a^{-1} yr^{-1})**	**C sequestration rates (t·CO_2e·ha^{-1} yr^{-1})**		
				0.1	1	3
Planned Riparian margins	3,200	4.5		0.45	4.50	13.51
Cable stream	1,260	0.756 (0.504)*				
Irrigation ponds						
Wetland	1,080	1.89		0.19	1.89	5.68
Total C sequestered (t·CO2·yr^{-1})		**7.151 (6.899)**		**0.64**	**6.40**	**19.19**
Emissions from farming operations (t·CO2·yr^{-1})		116.22 ha	8.5	989	989	989
Fraction (%) of GHG emissions by livestock				0.00	0.65	1.94

*Area that could be planted in trees

The scenarios attempt to represent the amounts of C sequestered over the first 10–15 early years as the trees gain size. The potential contribution of the proposed riparian planting programme for Scott Farm will make a very small contribution (c. 0 – 2%) to the C balance of the farm operation as a whole. A similar finding for the intensive sheep operation in Southland.

Figure H 3: Proposed riparian planting programme (outlined in light green) for DairyNZ Scott Farm mapped in the DairyNZ Riparian planner.

f) Case Study Summary

For the intensive sheep and dairy case study farms, both in lowland environments with very small areas in riparian margins, woodlots or shelter belts, sequestration by current and planned vegetation accounted for only a very small percentage (c. 0 – 2.5%) of current emissions by livestock.

For the sheep and beef case study farm operation in hill country, where there are significant areas of native bush, soil conservation plantings and woodlots, sequestration by current and planned vegetation accounts for a significant percentage (c. 5-20%) of current emissions by livestock. Biomass carbon stocks (e.g. standing forest, peat mires, soil carbon) of themselves do not create any offset benefit. Only change (increase) in those stocks results in an offset sink (sequestration). The native forest on the hill country sheep and beef case farm is an example of a C stock that is close to equilibrium.

Biomass carbon sinks are temporary (ending when the vegetation reaches a steady-state when growth and mortality/decomposition are balanced), while the many other co-benefits (water purification, shelter, erosion control, etc.) are on-going. In most cases here longevity will be <30 years. The age of the extensive shelterbelts on the intensive sheep farm case study located in Southland is a case in point.

The above analysis did not consider the influence of a change in vegetation type (e.g. pasture to forest) on the stocks of carbon found in the soil.

Before a farm considers increasing their biomass offset sinks they need to (i) have made an assessment of any outstanding environmental actions (erosion control, riparian plantings, wetland protection, shelter plantings, etc.) required to protect their natural capital stocks and farm operations and beyond that (ii) know the related cost/benefit of any further plantings.

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I. Drivers and Barriers (Steve Wakelin, Larry Burrows & John Quinn)

Recognition of sequestration sinks that are currently unrecognised in international and domestic accounting could benefit farmers and New Zealand as a whole. Farmers will be able to add offset C units into their farm emissions budgets, and New Zealand could do the same for its national C accounting.

Results from this study indicate the following potential sequestration rates for the vegetation classes reviewed above (Table I1).

Table I 1: Potential ranges of sequestration rates for the vegetation classes addressed in this report ($t \cdot CO_2e \cdot ha^{-1}yr^{-1}$). An estimated median sequestration rate is indicated after outlier top rates were excluded. Note that almost all these values originate from surrogate vegetation rather than data from the actual class and there is considerable uncertainty around all of them

Class	Sequestration rate range	Estimated median	Estimated Sequestration time period* (yrs)	Outlier rate in brackets omitted
Wetland, natural	0–2.0	2	>50	
Wetland, constructed	0	0	30	
Riparian	0–5.28 (15.3)	3	20	Dense gorse
Poles	2.0	2	30	
Shelterbelts	6.5–26.3 (28.8)	16	28	
Small lots	6.5–26.3 (28.8)	16	28	
Retired land	0–0.47	0.2	>50	

*Biomass sequestration is finite and rates are not linear or constant as the vegetation develops. As vegetation reaches maturity sequestration slows, or if vegetation is cleared/harvested (e.g. Small Woodlots) it becomes a source. These time periods are unconfirmed estimates.

If the potential extent of each of the classes is spatially depicted and expanded by these sequestration estimates in Table I1 they amount to a considerable unrecognised potential sink. Some classes (e.g. Riparian strips, Pole plantings, Small woodlots or Shelterbelts) are regionally widespread and abundant and already widely accepted as normal farming practises in some districts. Others at times represent large expanses of land that offer little obvious productive benefit to a farm but appear to hold significant carbon stocks (e.g. Retired land), while some (Riparian) are changing rapidly.

Significant carbon stocks may also be held in native vegetation (especially mature forest), wetland peat mires and soil organic matter. Although retention of these stocks may not result in direct economic benefit by ongoing quantifiable C sequestration, there can be many co-benefits to farms protecting these stocks. Examples of indirect drivers and co-benefits are listed in Table I 2. Guidelines and some regulatory protection of existing native forests and wetlands, and indirectly their carbon stocks, also exist in National Policy Statements, the Forests Act (1949) and Regional or District Plans.

Potentially large offset sinks could result from concerted national initiatives even though specific sequestration rates might be low. For example, Tanner et al. 2013 suggested a national wetlands trade-off area of 2.5% of productive farm area which could potentially sequester some C. Similarly, if 50% of streams and rivers adjoining farmland across all New Zealand had 10 m riparian strips established they would amount to >212,500 ha that might sequester 0.72 Mt $CO_2e{\cdot}yr^{-1}$. There is a high degree of uncertainty around these numbers, but they indicate the potential scale of the sink.

There are already drivers for establishing and enhancing wetlands and vegetation on farms outside consideration of carbon sequestration and many farmers are already engaged in this activity. There are also barriers that need to be overcome to achieve the full sink potential described and also to implement a system that allows these sinks to be recognised. These drivers and barriers are discussed in the following sections.

a) Drivers for establishing and enhancing wetlands and non-forest vegetation on farms

All of the on-farm activities addressed in this report provide numerous co-benefits to farmers in particular and society in general, and these 'ecosystem services' (Dymond et al. 2014) are being increasingly recognised. In many cases it is difficult to quantify or monetise these co-benefits because they may not be direct or immediate, but an indication of possible economic and societal benefits is given in Table I2. Non-economic benefits to landowners from each land use are generally those shared by society, including enhancements to biodiversity and landscape values. They can also provide a more diverse and pleasant working environment.

Table I 2: Drivers in the form of co-benefits attributed to the land-use practices addressed in this report

Land use	Economic benefit to landowner	Benefit to society, region or nation
Wetlands	Water quality Retain and transform contaminant inputs Reduce nitrogen export Flood mitigation	Water quality Biodiversity Landscape values Flood mitigation Wildlife habitat Summer streamflow maintenance
Riparian strips	On-farm flood mitigation Aquatic weed control – drainage Bank stabilisation (soil conservation) Livestock shelter/windbreaks Emergency fodder Water quality limit compliance Licence to operate Pollinators Decreased pugging of soils Reduced stock deaths Increased property value Reduced labour costs Good citizen	Water quality Stream temperature mitigation Nuisance instream vegetation control (shade) Biodiversity Landscape values Flood mitigation Fisheries
Pole planting	Stock feed, feed during drought Stock shade, thermal stress relief, welfare Erosion control, soil conservation	Erosion control Flood mitigation (sedimentation) Landscape values
Shelterbelts	Improved pasture productivity Improved stock welfare and productivity Improved irrigation efficiency (reduced evapotranspiration) Reduced topsoil erosion Reduced crop damage	Biodiversity Landscape values
Small woodlots	Timber products and revenue Shelter benefits Income diversification to improve business resilience	Biodiversity Landscape values Flood mitigation
Other non-forest revegetation	Livestock shelter/windbreaks	Biodiversity Landscape values

Carbon sequestration has not been a strong driver of on-farm vegetation management in the past, and interestingly still didn't appear as an issue for farmers in the Maseyk et al. (2017) study of farmers' views on riparian establishment. An example of an indirect or delayed beneficial effect is shown by the recent results from a study that shows a beneficial response to riparian establishment on water quality and aquatic ecosystems a decade or more later (Graham et al. 2018), which has indirect implications for adjacent landowners 'licence to operate'. Taken together, water quality, animal welfare and climate change mitigation provide drivers that encourage the land uses covered by this report. Currently this is mainly through market access, 'licence to operate' or marketing considerations, and in some cases through regulation (e.g. nutrient limits, clearance of native forest).

b) Barriers to changing farm management in general

Decision-making by rural land managers has been the subject of extensive research, often of a sociological nature based on theoretical frameworks of behaviour change, such as Universal Value Theory (Schwartz 2012), Theory of Planned Behaviour (Ajzen 1991) and Practice Theory (Bourdieu 1977). Drivers and barriers change over time, so it is difficult to predict future behaviour based on past experience, and research can become dated. For example, 'timber revenue' rarely featured as a reason for planting trees before the Asian market price spike of the early 1990s, but it was a dominant reason given in later surveys. Few farmers planting in the early 1990s would have been aware of the potential for revenues from carbon trading (Wakelin et al. 2014).

MPI commissioned a recent report on the current drivers and barriers to changing land use in New Zealand (Journeaux et al. 2017). The report suggested that the strong economic incentives to optimise land use are insufficient to overcome the barriers. Key barriers were found to be:

- i land conversion costs
- ii farming preferences
- iii information deficit.

A review of the effectiveness of the Sustainable Land Use Initiative (SLUI) found that both SLUI farmers and those hill country farmers who have chosen not to sign-up to SLUI identify funding and time as the primary barriers to any speeding-up or extension of their environmental efforts (Brown et al 2016).

The 2015 Survey of Rural Decision Makers received the responses in Table I3 as reasons for not changing or intensifying land use, or increasing farm size.

Table I 3: Reasons given for not changing or intensifying land use, or increasing farm size

	Frequency	% respondents
Lifestyle decision	252	53.6
Anticipate retiring soon	59	12.6
Lack of financing	48	10.2
Environmental decision	42	8.9
Other	69	14.7

The 'Other' category contained a wide range of reasons, many also relating to lifestyle (e.g. age, already retired, don't want to change, happy as I am).

More light was shed on lifestyle barriers by a Motu report that looked at barriers to 'no-cost mitigation options', where the options were assumed to be those that reduced the adverse environmental impact of a farm without reducing farm profitability, as measured in conventional financial terms (Jaffe, n.d.). Barriers were grouped into seven categories:

1 efficient (non-barriers) – where the simple financial profitability test has failed and a rational financial decision has been made
2 information deficit
3 market/institution failure
4 externalities (e.g. costs or benefits shared beyond the decision-maker)
5 regulatory/policy barriers
6 risk and uncertainty – either rational calculation of true financial benefit under risk, or failure to assess true risk (the former is category (1); the latter is category (g) (Jaffe))
7 behavioural barriers – cognitive bias (the examples given generally respond to elements of the theories of behaviour change).

The literature suggests that a financial incentive in the form of revenue from C sequestration is unlikely, by itself, to lead to land-use change to the land uses investigated in this report: the ETS has had limited success in encouraging afforestation (Evison 2016). Land use change may still take place because of the other benefits seen by farmers that they do not associate with ETS-compliant forests.

c) Barriers specific to establishing and enhancing wetlands and non-forest vegetation on farms

There are barriers to implementing each of the land-use changes investigated in this report, many of them in common. In broad terms, a landowner will not implement the change if they do not perceive benefits or if there are no benefits.

- No perceived benefits:
  - information deficit (e.g. the real costs, benefits and risks are not well understood)
  - cognitive bias (e.g. not a good fit with landowners' values)
  - risk (e.g. there is uncertainty around productivity, prices, carbon price, regulations, future land-use choices, etc. and the landowner is risk-averse)
- No actual benefit to landowner:
  - poor fit with operations (e.g. irrigators)
  - inflexibility (e.g. long investment horizon)
  - time and money investment not justified.

In general, costs are often tangible, direct, immediate and known, while benefits are intangible, uncertain and deferred. In these circumstances, taking on debt may be unacceptable, even if the return on investment appears to be excellent. There is also value in deferring decisions, as the regulatory environment may change. For example, even though shelterbelts can be shown to provide direct economic benefits (see Table I2), there are still a number of adverse impacts that must be taken into account on a case-by-case basis, including:

i the cost of fencing, establishment and maintenance

ii loss of production land

iii competition for water and nutrients with crops and pasture

iv time required

v impeding irrigation

vi creation of frost pockets

vii shading of tracks, preventing them from drying out or thawing

viii impeding line of sight, preventing casual monitoring of stock

ix boundary disputes, power lines, Kiwirail, civil aviation regulations

x damage to fences from falling branches

xi harbouring pests and weeds.

d) Barriers to inclusion within the ETS or equivalent scheme

The ETS is structured to be broadly consistent with international accounting. This means that it shares common distinctions, such as:

i rewards are for changes in activity (additionality), not business-as-usual management

ii maintenance of forests established before 1990 (Baseline) is assumed to be business as usual, whereas forests established after that time are eligible for credits

iii principles of transparency, accuracy, consistency (including symmetry), comparability, completeness, verifiability and efficiency apply to reporting.

New Zealand carbon units (NZUs) devolved to participants in the ETS are obtained through New Zealand's accounting under international agreements. If the ETS is expanded to capture more land uses, it would no longer be consistent with international accounting. If the total number of units allocated is small, this may not matter because not all eligible forest has been entered into the ETS. However, there could be issues with land that transitions between new 'non-forest' ETS classes and forest – in either direction.

If international reporting is expanded to include new categories, there would be issues with mapping and measurement at the national scale. Mapping the existing land-use sub-categories is difficult enough, but the advantage is that a high proportion of the total C stock is in a stable land use and can be captured with reasonable certainty. Mapping and measuring small areas in changing landscapes with low C stocks would be much more difficult. Natural variability creates uncertainty, and with small stocks and stock changes it is difficult to be confident about the direction of change given the large error associated with such estimates.

Remote assessment and delineation of fuzzy boundaries or boundaries that change between weather events (e.g. wetlands), and probably other land-use classes, are difficult for remote image classification systems to identify due to their genuinely variable size, shape and phenology (Bellis et al. 2017). On-ground definitions of the boundaries of wetlands will be complicated due to seasonally ephemeral wetlands and flooding events. Shelterbelts were initially delineated in LCDB but are now not included due to difficulties in definition (i.e. what is a shelterbelt?), differentiation of species and separation from surrounding cover, and scale (LCDB maps areas of >1 ha, therefore a 5 m wide shelterbelt would have to be over 2 km long to be classified as a shelterbelt). Hence it is difficult to determine a useful national baseline for this class.

At the farm level, implementation would need to allow look-up tables, because the cost of verification through a field measurement approach (as used for forests) is likely to outweigh the returns – just as it currently does for forests below a minimum break-even size. Only about 60% of eligible post-1989 plantation forest and a smaller proportion of eligible reverting indigenous forest have registered in the ETS, which suggests there may be limited uptake for non-forest vegetation types given their lower expected sequestration potential. The cost:benefit ratio would change with a much higher carbon price, but there would be issues with potential liabilities due to maintaining the C sink, insurance costs and natural disturbances.

Assuming that vegetation classes corresponding to the lookup tables can be defined, there are no major technical barriers to landowners mapping and classifying areas of vegetation on their own properties. MPI provides an online mapping tool for use with the ETS and farmers can supplement aerial photography and satellite imagery with ground-truthing based on their in-depth local knowledge. Baselines may still be difficult to establish as landowners may not necessarily know the full land use history including vegetation establishment years.

While the use of lookup tables would lower participation costs, there is a need for the system at a farm scale to maintain credibility, and the overall cost of mapping, measuring, quantifying, registration, auditing and monitoring permanence of small or scattered vegetation would be high and estimates would carry high uncertainty.

Similar issues apply to the recognition of C units by 'grey markets' outside the ETS. Independent verification standards and the perceived quality of units for C uptake, biodiversity and landscape value enhancement due to these on-farm activities affect their value to the wider market, as long as there are willing buyers. Farming for C alone is unlikely to fetch any market premium. To obtain the highest recognition and value for C units requires the highest verification standards to be met, and an attitude to treat C as a part of a broader land management approach is required (see Burrows et al. 2018). Such a market needs the credibility provided by the same processes built into the ETS. Without this, there is the risk of a consumer backlash.

Further complications that act as barriers to landowners are the disadvantages in having land locked into a scheme (permanence is an important component of such schemes), the potential liability for past and future clearance of these vegetation types, and the Baseline date from which change is determined.

Having land locked into any offset scheme and utilising C units in any way creates a future liability against the property, because the landowner needs to ensure that the physical stocks equivalent to units transferred are permanently retained on-site. That affects future land-use options and the potential value and/or saleability of farmland, as the offset units are tied to the land in the case of the ETS.

The question around a Baseline date means that from some fixed point of time all net GHG emissions changes, either incoming or outgoing, will need to be accounted for and balanced. That also has implications for future land-use options. For example, if a shelterbelt was established on a property, it would create sequestration units as it grows, but if another shelterbelt on the property was cleared as part of ongoing farm development decisions, or it was flattened due to an unforeseen weather event, then the net loss of carbon would also need to be accounted for. Likewise if the net area of wetlands continues decreasing on farmlands nationally there will be a net loss of C on average, even if some farms with peat mires see a C benefit and protect them.

Finally, all of the on-farm potential sequestration land-use practices addressed in this report have identified inadequate New Zealand data and information on emissions factors as a major barrier to their use. Surrogate emissions factors derived for other vegetation classes do not meet the IPCC (2006) guidelines of best practice and the need for unbiased factors, and fall short of providing the credibility needed if they are to be used in a look-up table approach for a domestic scheme. Resolving New Zealand emissions factors for each class and developing a landowner system for mensuration, assessment and monitoring will be a major undertaking. The first steps to providing specific emissions factors for these offsets will be to commission design studies to quantify the resources.

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6 Conclusions

Most of the vegetation classes considered here (riparian establishment, pole planting, shelterbelts, and small woodlots) are widespread and common on farmland in some regions of New Zealand, are already accepted practices, and as such could contribute GHG sequestration benefits to New Zealand farms. In practise the intensive sheep and dairy farm case-study examples investigated indicate that offsets from these vegetation classes in these two cases account for only a minor proportion (0 – 2.5%) of overall emissions from these farms under current practises. For the sheep and beef farm case-study investigated offsets from these vegetation classes accounted for a significant proportion (c. 5-20%) of overall emissions under current practice.

In some cases (woodlots, shelterbelts, riparian) and on some landscapes (e.g. hill and steepland) there is potential for increased sequestration, but that is limited unless it is associated with a major targeted initiative and hinge on favourable cost/benefit ratios. Including them in future C inventories would seem a logical step. However, realizing any potential will require quantifying spatial extent, sequestration rates, longevity of the sink, defining a time baseline, and adoption into a New Zealand scheme, while addressing actual and perceived barriers by farmers. This will not be simple. Only a deliberate, additional national initiative aimed at significantly increasing the land area of those classes will realise any sizeable C sequestration.

Most of the time biomass carbon stocks (e.g. standing forest, peat mires, soil carbon) of themselves do not create any new offset benefit. Only net change (increase) in those stocks results in an additional offset C sink (sequestration). Measurement of that rate of change is critical to realising any potential sink. Also, biomass carbon sinks are finite (the sink saturates once growth and mortality/decomposition are balanced and the vegetation attains a steady-state condition). Which is why measuring and accounting for baseline stocks, rates of change, spatial extent and longevity of the sink are critical to realising any potential sink, and for subsequent implementation.

The above appraisal shows that actual sequestration rates, longevity of the sink and temporal baselines specific for all the vegetation classes are unknown. Here we have used surrogates from similar vegetation with some known sequestration rates. Setting objective baselines and identifying conditions at that temporal starting point will not be simple.

There is a deficiency of data and statistical estimates of carbon stocks and sequestration potential for all non-ETS compliant on-farm units considered in this report. Some surrogates of carbon stocks and sequestration potential used here have been taken from related contexts but are expected to be only partly comparable and not fully adequate (e.g. riparian strips host different vegetation structures and growth conditions compared with typical successional indigenous forest). In addition, there are no adequate estimates of spatial extent and change of these vegetation classes on New Zealand farmland as a whole, and as yet no standardised means of quantifying them.

The multiple conditions and complexities associated with each of these units (e.g. major differences between mires/peatlands, marshes and constructed wetlands, or, between shelterbelts of different species and whether or not they are hedged/mown) imply a large level of uncertainty and wide range of variability in their C stocks and sequestration potential. This variability and uncertainty could be reduced with the specification of particular conditions (e.g. peatlands instead of wetlands).

Knowledge gaps can be readily addressed by future research and we recommend for any potentially viable classes that the first step is to quantify actual emissions factors for these offsets by undertaking specific design studies. We note that related questions have been addressed for some time and (i) will be methodologically challenging (e.g. tracing the flow of various biomass compartments below-ground or in water), (ii) may require time to be answered given the pace of some natural processes or use of a space-for-time approach, and (iii) may pose challenges for implementation and monitoring (e.g. what land area ought to be assigned to shelterbelts if stocks and sequestration estimates originate from continuous forest?).

The on-farm classes considered in this report contribute considerable associated co-beneficial farm, ecosystem, and biodiversity services beyond carbon sequestration. They are already identified - at least to some extent - in farm environmental plans and it would be technically feasible for a motivated farmer to map and classify the vegetation on their own land. Cost-effective verification and monitoring would be more difficult.

Next steps towards development of comprehensive emissions factors

Resolving New Zealand emissions factors for each class and developing systems for mensuration, assessment and monitoring on individual farms will be a major undertaking. Three main objectives are envisaged for each vegetation class deemed to offer significant sequestration potential. i) Estimate the extent of the resource through time on farmland across New Zealand, ii) Develop comprehensive emissions factors that account for variation within the vegetation class across the country and through time, and iii) Develop mensuration techniques that can be used to assess the biomass and sequestration of each class on farmland. Objective ii) would form the basis for development of appropriate lookup tables.

To quantify emissions factors for these offsets will require commissioning specific design studies. Methods would likely include combinations of indirect or remote image-analysis techniques, gas-exchange methodologies and modelled results will need to be informed and validated by representative ground-based sampling. This latter would also provide the basis for development of on-farm assessment and monitoring methodologies. Each vegetation class will have its own combination of approaches and need to accommodate regional and environmental variation, and understand changing EFs during development of the offset through time to maturity using repeat measures in some instances where previous sampling may exist, or space-for-time studies where it doesn’t.

7 Recommendations

There is a certain amount of inertia for any change in land-use practises but an important and useful driver of change is the economic one. Before any farmer can consider increasing their biomass offset sinks they need to know the associated cost/benefit. What is the cost of production of C-units relative to livestock units? We recommend that an economic analysis is carried out and it considers the area of farmland that would be needed to offset typical farm emissions. A number of land-use options, sequestration rates and sink longevities, such as those presented here, would need to be clearly specified in such an analysis.

It is recommended that the results presented here for the on-farm vegetation classes are considered in relation to the likelihood of any of these classes being adopted as an expanded, additional, significant new C offset sink on farms beyond business-as-usual land-uses.

Most of the vegetation classes addressed in this report have potential to sequester significant C, yet there is a deficiency of data and statistical estimates of actual carbon stocks, potential sequestration (and variability) and longevity of the sink for all non-ETS compliant on-farm units considered in this report. For these knowledge gaps to be filled for the sake of both New Zealand farms, and national accounts, will require several specific quantitative investigations of actual extent and sequestration to be commissioned that may be associated with existing research programmes.

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