Plantation forestry: Review of baseline performance, source control and device management options for Freshwater Management Tool

Prepared for

Auckland Council

Report prepared by

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August 2022

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Status:	FINAL	

Bibliographic reference for citation: Muller, C. 2022. Plantation forestry: Review of baseline performance, source control and device management options for Freshwater Management Tool. Final report for Auckland Council. 91 pages.

Executive Summary

Auckland Council (AC) is developing a continuous, process-based catchment model (the Fresh Water Management Tool, FWMT) to inform its operational and regulatory decision-making for water quality. Scenario capability in the FWMT requires accurate classification of land use activities for their differences in ongoing management, and knowledge of potential mitigation opportunity, cost and effect tailored to the activity and catchment.

This report forms a part of the ongoing improvement of the decadal FWMT programme. Specifically it reviews forestry literature, focusing on plantation forestry, to try and quantify base economic and environmental footprints as well as the opportunity and impact of activities to mitigate contaminant loss, for example, alternative harvesting methods. The FWMT uses four forestry groupings; Farm Forest (both native and exotic forestry), Urban Tree, Mixed Forest and Plantation Forest. Data is typically available for permanent forest, plantation forestry and farm forestry (plantation forestry within a pastoral title).

The National Environmental Standard for Plantation Forestry (NES-PF) provides a consistent national standard to manage forestry activities, meaning that the majority of forestry activities across the country operate as permitted activities subject to compliance with the regulations in the NES-PF. The NES-PF could form the basis for defining the starting point or 'base state' in the FWMT from which mitigation is applied. Alternatively, it could form the basis for the first mitigation bundle (i.e., as a proxy for good management practice). From this point the New Zealand Environmental Code of Practice for Plantation Forestry would be considered a further mitigation bundle or best management practice. However, more information is required on current forestry practices across the Auckland region to help define the base state and opportunity for further mitigations to be applied.

Determining the base environmental impact of forestry, as well as determining the opportunities, costs and benefits of alternative plantation forestry practices (source controls) is very challenging. Markedly less research has occurred in New Zealand into forestry system design and management choices for water quality effect than much of the pastoral and horticultural sectors. Where research does exist on the environmental impact of forestry it is largely focused on comparing forestry with alternative land uses (e.g., pastoral) rather than comparing management practices within forestry. In addition, forestry is challenging to incorporate into the FWMT due to the variation in economic and environmental impacts across the production forestry cycle.

Pinus radiata is the most common plantation tree in Auckland and is typically harvested at 27 years. As such, the current base state is based on this forestry system for both the farm and plantation forestry groups, as well as permanent native forest. It is recommended that at this stage the FWMT uses an annualised profit estimate which enables data to be included into the current FWMT format. In future iterations it is suggested that specific forestry modelling is undertaken to capture economic impacts of mitigations and allow more flexibility in adjusting profit across the typically 27-year plantation cycle to the 50-year life cycle cost (LCC). While it would be desirable to generate LCC based on forestry modelling that is specific to the Auckland region and underpinned by discussions with Auckland foresters, the current annualised profitability estimates are \$185/ha/yr for farm forestry (plantation), \$430/ha/yr for plantation forestry and \$500/ha/yr for permanent forestry (noting this is very sensitive to carbon prices). These are based on national estimates so while the data can be applied to Auckland, it is not Auckland specific.

It is challenging to assess the environmental footprint of forestry, especially across key parts of the forestry cycle such as the window of vulnerability. Nitrogen losses can be separated out across key phases in the forest cycle for farm, plantation and permanent forest. Sediment losses can be separated out across key phases in the forest cycle for farm and plantation forest. However, P losses are unable to be identified across the forestry cycle from the literature (noting a benefit of process-modelling is that if P-loss is governed or linked to erosion, then the erosional rate estimates for sediment in the FWMT can be used to guide how P loss varies over the forest cycle). In terms of contaminant footprint, it is assumed that farm and plantation forestry have the same base footprint. Plantation forestry (and farm plantation forestry) are assumed to have an annual base footprint of 2.5 kg N/ha/yr, 0.2 kg P/ha/yr and 87 5kg sediment/ha/yr averaged across the rotation. Permanent forestry is assumed to have an annual footprint of 3 kg N/ha/yr, 0.1 kg P/ha/yr and 875 kg sediment/ha/yr.

Various mitigations are assessed in the literature. However, these are often discussed in qualitative not quantitative metrics. In addition, forestry is often compared to alternative land uses rather than comparing mitigations within forestry (i.e., improved practices and devices). Mitigations that are discussed in literature but are not able to be quantified due to a lack of information on cost, benefit and/or opportunity were harvesting methods, post-harvest vegetation management, sediment control measures, fertiliser use and wetlands. Riparian areas can be quantified in terms of cost (lost profit) and there are some quantified estimates of the environmental impact, especially for 5 m riparian areas. However, some literature suggests that there is no difference in the sediment performance of buffers greater than 5 m wide. There is also a challenge in assessing the opportunity for riparian buffers to be applied relative to current practices.

This report recommends next steps to fill the forestry knowledge gap for the FWMT programme. Initially this includes alignment of the forestry hydrological response units (HRUs) in the FWMT with the groups for which data exists, namely permanent forestry, plantation forestry and farm plantation forest, incorporating the base footprints identified in this report into the FWMT and incorporating riparian buffers as an initial mitigation into the FWMT. Three key areas of further work are recommended if forestry is to be further integrated into the FWMT. First is developing a model to generate profitability metrics for the different HRUs that are specific to the Auckland region and allow for the economic impact of mitigation options to be assessed. Secondly, it would include working with experts to confirm base footprints as well as estimating the environmental impact of mitigations. Thirdly, to enable mitigations to be applied in the FWMT, a better understanding is needed of the current state of practices in order to inform the opportunity for these mitigations to be applied across forestry HRUs.

CONTENTS

List of Acronyms

AC	Auckland Council
FWMT	Freshwater Management Tool
N	Nitrogen
P	Phosphorus
NES-PF	National Environmental Standard- Plantation Forestry
NPS-FM	National Policy Statement for Freshwater Management
E coli	Escherichia coli
HRU	Hydrological Response Unit
LCC	Life Cycle Cost
NFED	National Exotic Forest Description
ESC	Erosion Susceptibility Classification
NZLRI	New Zealand Land Resource Inventory
LUC	Land Use Capability
BEP	Best Environmental Practice
GMP	Good Management Practice
The Code	New Zealand Environmental Code of Practice for Plantation Forestry
NPV	Net Present Value
IRR	Internal Rate of Return
EAA	Equivalent Annual Annuity
CBA	Cost-Benefit Analyses
FIF	Forestry Investment Framework

1 Introduction

Auckland Council (AC) is developing a continuous, process-based catchment model (the Fresh Water Management Tool, FWMT) to inform operational and regulatory decision-making for water quality. The FWMT is a freshwater accounting system, able to report changes in hydrology and contaminant generation and loss, instream and downstream, for both urban and rural land throughout the Auckland region. The FWMT includes capability to determine "current" or baseline state of contaminants, as well as "future" or scenario state of contaminants under changes in resource management (land cover, intensity of use, water consumption and discharge to water) as well as climate.

Scenario capability in the FWMT requires accurate classification of land use activities for their differences in ongoing management, and knowledge of potential mitigation opportunity, cost and effect tailored to the activity and catchment.

This report forms a part of the ongoing improvement of the decadal FWMT programme. Earlier reporting for the FWMT Stage 1 has reviewed and developed information on rural sector water quality management options (e.g., opportunity, cost and effect). FWMT Stage 1 management reporting includes:

- Muller et al. (2020a) provided a literature review of the efficacy of the range of primary sector responses to lower the contribution of key water quality contaminants and their accompanying economic impacts.
- Muller and Stephens (2020a) provided detailed information on incorporating riparian area management scenarios into Stage 1 of the FWMT.
- Muller, Ira and Stephens (2020b) translated the literature estimates of cost and efficacy into a Life Cycle Cost (LCC) Model for application to the FWMT Stage 1.
- Muller and Stephens (2020b) highlighted areas for refinement of future FWMT stages in relation to the rural sector mitigations (e.g., choices of, cost and benefits for, granularity of across contaminant sources and regionalization for Auckland farm systems).

Earlier FWMT Stage 1 management reporting is limited to pastoral and horticultural productive options. This report complements earlier reporting by reviewing management options available for plantation forest activity (on FWMT Stage 1 contaminants), to explore and recommend how management choices can be configured in future FWMT development. Specifically, this report provides:

- Requirements of plantation forestry under the National Environmental Standard for Plantation Forestry (NES-PF) to manage environmental impacts, and relevance to FWMT modelling
- Information on the base (current state) profit and/or water quality impacts associated with plantation forestry
- Knowledge of actions suited to plantation forestry mitigation of water quality effects

Environmental impacts and mitigation actions are considered here for impact of plantation forest on nitrogen (N), phosphorus (P), sediment and *Escherichia coli* ("*E coli*") discharge to freshwater. It is focused on discharge to water rather than instream or coastal effects. We review cost and benefit data from various studies and information and then note key considerations governing application in the FWMT.

At the outset, determining the opportunities, costs and benefits of alternative plantation forestry practices (source controls) is especially challenging. Markedly less research has occurred in New Zealand into forestry system design and management choices for water quality effect than much of the pastoral and horticultural sectors. Few paired studies exist comparing alternative management actions within the plantation forest sector, for effects on water quality or lifecycle cost.

1.1 Report Structure

This report is structured as follows:

- The remainder of Section 1 sets out the background to the FWMT, how forestry fits into the FWMT and key information on forestry in the Auckland Region.
- Section 2 highlights the current policy requirements and industry recommended practices for plantation forestry. The section is focused on how these current policy requirements and industry standards help define the opportunity for device and source control practices across forestry in the FWMT as well as contributing to the understanding of the base footprint.
- Section 3 reviews the literature on the base state of forestry, both economic and environmental. This is currently intended for use in supporting the calibration of the FWMT, although consideration is given to its suitability for use in future scenario modelling.
- Section 4 reviews the literature for potential device and source control mitigation options for forestry.
- Section 5 provides recommendations for the inclusion of forestry int the FWMT at this stage of FWMT development. Recommendations cover the base state (economic and environmental), mitigation options and their associated opportunity. It also highlights where improvements could be made in further iterations of the FWMT for forestry.

1.2 Background to the FWMT

The FWMT simulates hydrology and contaminant response of land to climate and resource use, by classifying the Auckland region into unique biophysical and land use types – so-called Hydrological Response Units (HRU), each representing how hydrological and contaminant processes respond differently to variation in climate, across 490,000 ha of land.

HRU classes are defined by combinations of land cover, intensity of use, hydrologic soil group and slope. HRU composition has been assessed for 5,465 sub-catchments, to define a "static" baseline of landscape within the FWMT Stage 1. The landscape has been configured to represent the varying state of land use within, and across, sub-catchments over the 2013-2017 baseline period but, being static, does not incorporate ongoing changes in land use through that period.

Each HRU is uniquely parameterised for hydrological and contaminant processes, on a regional basis in the FWMT (i.e., land titles of equivalent class, under identical climate, are assumed to generate identical contaminant loads in equivalent runoff, interflow or active groundwater). The development of the HRU framework, including all sources of data and transformation, is detailed in the Baseline Inputs and Baseline Configuration & Performance reports (see Auckland Council, 2021a and 2021b).

The cost and benefit model within the FWMT is based on a Life Cycle Cost modelling approach described in Ira et al. (2020). Only a high-level overview is provided here. LCC are used by the FWMT to identify least-cost long-term management actions for targeted changes in water quality—using continuous, process modelling to support "dynamic, cost-optimised intervention" modelling. The latter describes the FWMT Stage 1 capability to continuously simulate, on a 15-minute time-step, changes in the hydrology (runoff, interflow, active groundwater) and contaminant (build-up, wash-off, transport) responses of parcels of land subject to source controls, and of devices able to intercept and treat upstream discharges. The FWMT Stage 1 can identify least-cost actions over a 50-year discounted period, from feasible options, to alter flow and/or contaminant time-series. Importantly, the FWMT

Stage 1 requires information on the opportunities upstream of assessment points for management actions. Hence, three key sources of information are required for cost-optimised dynamic intervention modelling: opportunity (locations and extent of all potential actions), costs (50-year lifecycle) and effects (changes to hydrological and contaminant processes).

LCC include the sum of acquisition and ownership costs of an asset over its life cycle from design, construction, usage, and maintenance through to renewal or disestablishment (Figure 1). A cradle-to-grave time frame is warranted for the FWMT Stage 1 use as a catchment management tool, because future costs associated with a mitigation measure are often different than initial acquisition cost, and may vary significantly between alternative solutions (e.g., between grey and green infrastructure – Australian National Audit Office, 2001). Where the responsibility to pay these costs falls also often changes throughout the full lifecycle of the intervention. Use of LCC permits a long-term basis of decision-making over the full lifespan of investment services (i.e., prevents inefficient investment in the long-term). It also suits the nature of forestry which is not easily represented realistically on an annual basis like some other rural sector land uses; however, the 50-year time frame does not readily align with the prevalent rotation length in plantation forestry (ca. 27 years).

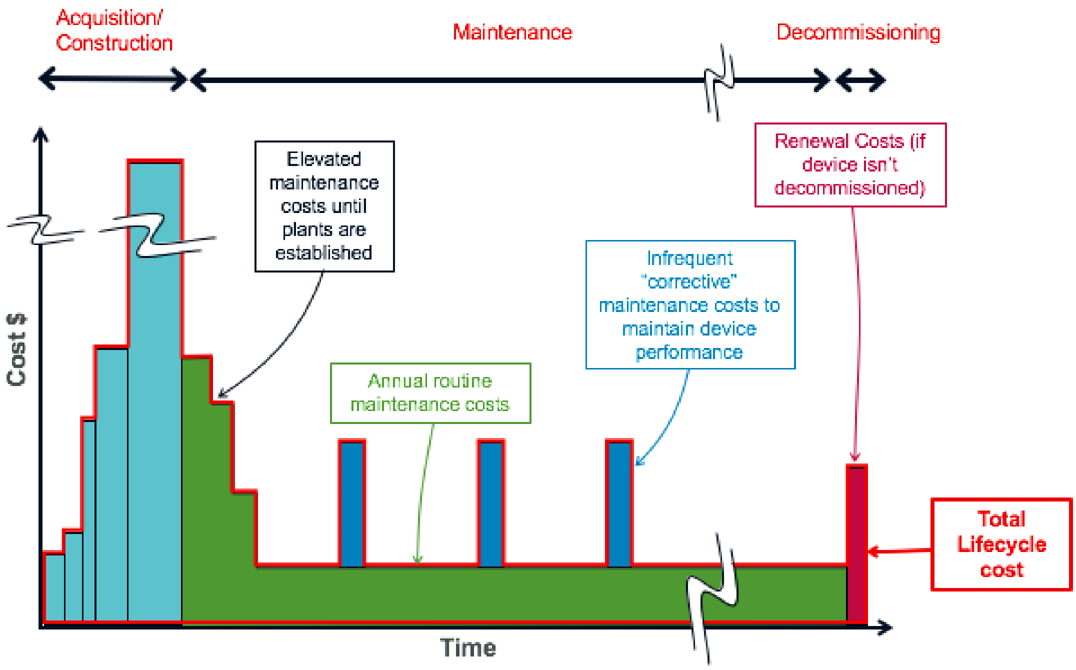

Figure 1: Phases in the life cycle of stormwater interventions and potential long-term costs (Ira et al., 2020)

A robust LCC model has been developed in general accordance with the Australian/New Zealand Standard (4536:1999) for LCC. The structure of LCC models is equivalent for all mitigations (across rural and urban sectors), with key assumptions made for FWMT Stage 1 including (but not limited to):

- A 50-year period has been used for costing to provide consistency with urban interventions (e.g., support integrated modelling of urban and rural water quality management in line with NPS-FM requirements – Policy 4 [MfE, 2019]).
- LCC are available for 2%, 4% and 6% discount rates, as recommended by Auckland Council's Chief Economist Unit (Ira et al., 2020).

- Base date for all LCC is 2019 and costs are New Zealand dollars (e.g., capital, maintenance, operating profit or opportunity cost).
- LCC exclude goods and services tax (GST).
- Total acquisition cost (TAC) includes an overhead and indirect cost factor of 17.5% of the construction cost – accounting for time needed to plan, consent or implement potential mitigations, and equivalent with overhead costs for urban interventions of 15% - 20% [Ira and Simcock, 2019]). TACs are only included with edge of field (EOF) and land retirement rural options (i.e., not included with bundled mitigations as these lack capital costs).
- Construction costs are allocated in the first year of the LCC model with renewal costs included as appropriate in future years. Maintenance costs are allocated for all other years. For rural options, costs such as opportunity cost (from retiring land in perpetuity) or reduced operating profit can be considered annually.

Annualised output from the LCC model offer indicative costs; variation can be expected in those for mitigations when applied but whose central cost should be similar. Hence, comparative accuracy will be far greater than accuracy in absolute cost, supporting use in optimisation assessments. LCC allows "like for like" comparison of the full spectrum of costs between mitigations (e.g., outlay, maintenance, opportunity or profit cost). However, LCC assessments do not include practical considerations around factors such as the feasibility, timing, uptake or optimisation of interventions in specific location(s), or about financing, governance or distributions of costs for particular catchments or activities.

1.3 Forestry in the FWMT

Forestry in the FWMT Stage 1 was initially categorised into "Forest High Impact" or "Forest Low Impact" HRU classes where land cover/use information indicated the presence of forest and where vegetation height was greater than 0.5 m (see Table 8–10 in Auckland Council 2021a for a full accounting of data sources and the process used to classify the forest HRU type). As the FWMT seeks to continually improve the underlying data that drives it, re-classification of forested land occurred under FWMT Stage v1.2 for reporting purposes. Revised reporting classification of forested land in the FWMT Stage v1.2 which then informed assignment to either the "Forest High Impact" or "Forest Low Impact" HRU classes is as follows:

- **Farm Forest** – areas on a pastoral and horticulture title (as per AsureQuality, 2016) with > 1 ha contiguous vegetation cover > 2 m in height. (Assigned to Forest Impact 1 HRU class.)
- **Urban Tree** – classified as urban as per rural-urban boundary (RUB) and ≤ 1 ha of contiguous vegetation > 2 m in height (both exotic and indigenous species). (Assigned to Forest Impact 1 HRU class.)
- **Mixed Forest** – if urban (as per RUB), where land cover is 'Exotic Forest' (LCBD5, 2020) or where > 1 ha of contiguous vegetation of > 2 m height. If rural (as per RUB), where vegetation > 2 m in height, exclusive of pastoral and horticultural titles (as per AsureQuality, 2016). (Assigned to Forest Impact 1 HRU class.)
- **Plantation Forest** – Rural 'Exotic Forest' and 'Forest Harvested' extent (RUB; Manaaki Whenua Landcare Research, 2020) and > 10 ha contiguous, excluding farm forest. Contiguous forested areas ≤ 10 ha assigned to Farm Forest or Mixed Forest. (Assigned to Forest Impact 2 HRU class.)

Based on the reprocessing of the forestry HRUs, Table 1 below indicates the approximate areas of each of the forestry HRU reporting categories in the base state of the FWMT v1.2 (Auckland Council, 2022).

Table 1: Area of each forest HRU reporting category in base state (FWMT v1.2)

Land cover	Definition	Area in base state (ha)
Farm Forest	Areas on a pastoral and horticulture title (as per AsureQuality, 2016) with > 1 ha contiguous vegetation cover > 2 m in height	47,688
Mixed Forest	If urban (as per RUB), where land cover is 'Exotic Forest' (LCBD5, 2020) or where > 1 ha of contiguous vegetation of > 2 m height. If rural (as per RUB), where contiguous vegetation > 2 m in height, exclusive of pastoral and horticultural titles (as per AsureQuality 2016).	77,915
Urban Tree	Where land is classified as urban as per rural-urban boundary (RUB) and ≤ 1 ha of contiguous vegetation > 2 m in height (both exotic and indigenous species).	8,297
Plantation Forest	Rural 'Exotic Forest' and 'Forest Harvested' extent (Manaaki Whenua Landcare Research, 2020) and > 10 ha contiguous, excluding farm forest. Contiguous forested areas ≤ 10 ha assigned to Farm Forest or Mixed Forest.	37,647

In terms of this report, the following forestry groupings are considered:

- **Farm plantation forest,** which has a unique profit but the same environmental performance as plantation forest. This is based on *Pinus radiata,* which is the majority of plantation forest in the Auckland region.
- **Plantation forest,** which has a unique profit but the same environmental performance as farm forest. This is based on *Pinus radiata,* which is the majority of plantation forest in the Auckland region. This includes timber and carbon (using the averaging approach meaning this area needs to be replanted after harvest) given the availability of data.
- **Permanent forest,** which has a unique environmental performance (that excludes harvesting impacts) and unique profit (essentially only carbon income as no timber production). It is suggested that this is applied to the Mixed Forest HRU, although it is acknowledged that not all Mixed Forest will meet the requirement for carbon income, and this should be considered when interpreting results. While permanent forest can be considered in terms of base footprint, no mitigations (either device or source control) are applicable as permanent forest is not actively managed for water quality benefits after establishment.

The Urban Tree HRU is not explicitly considered here as there is unlikely to be any profit from this (not eligible for carbon as less than 1 ha and not harvested for product) and there is limited quantification of environmental performance in this context.

The differences in the forestry HRU reporting groups and the groupings used in this report are described in **Error! Reference source not found.**.

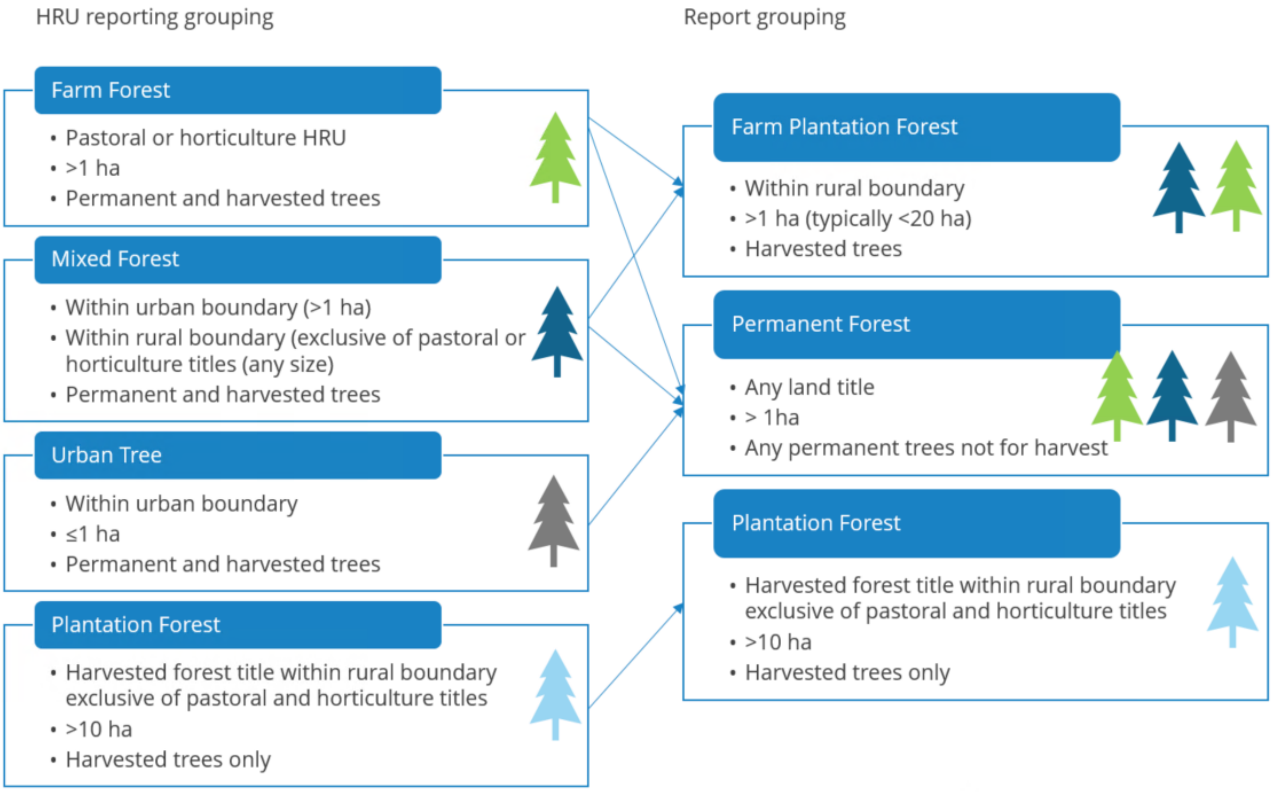

Figure 2: Relative forestry groups between HRU and this report

Forestry on farm has, in reality, both permanent and plantation forest subsections, although at a smaller scale than that in the plantation and permanent forest groups. The cost and environmental footprint of permanent farm forestry will be similar to the permanent forest HRU, just scaled appropriately. However, farm plantation forestry (i.e., plantation forestry) is likely to have a different cost profile due to economies of scale and therefore warrants a separate group to plantation forestry at this stage. In future iterations of the FWMT separating farm forestry into permanent and plantation subsections should be considered.

1.4 Forestry the Auckland Region

Information on the forestry sector typically separates the country into wood supply regions, Auckland falls into the Northland wood supply region which is shown in Figure 3.

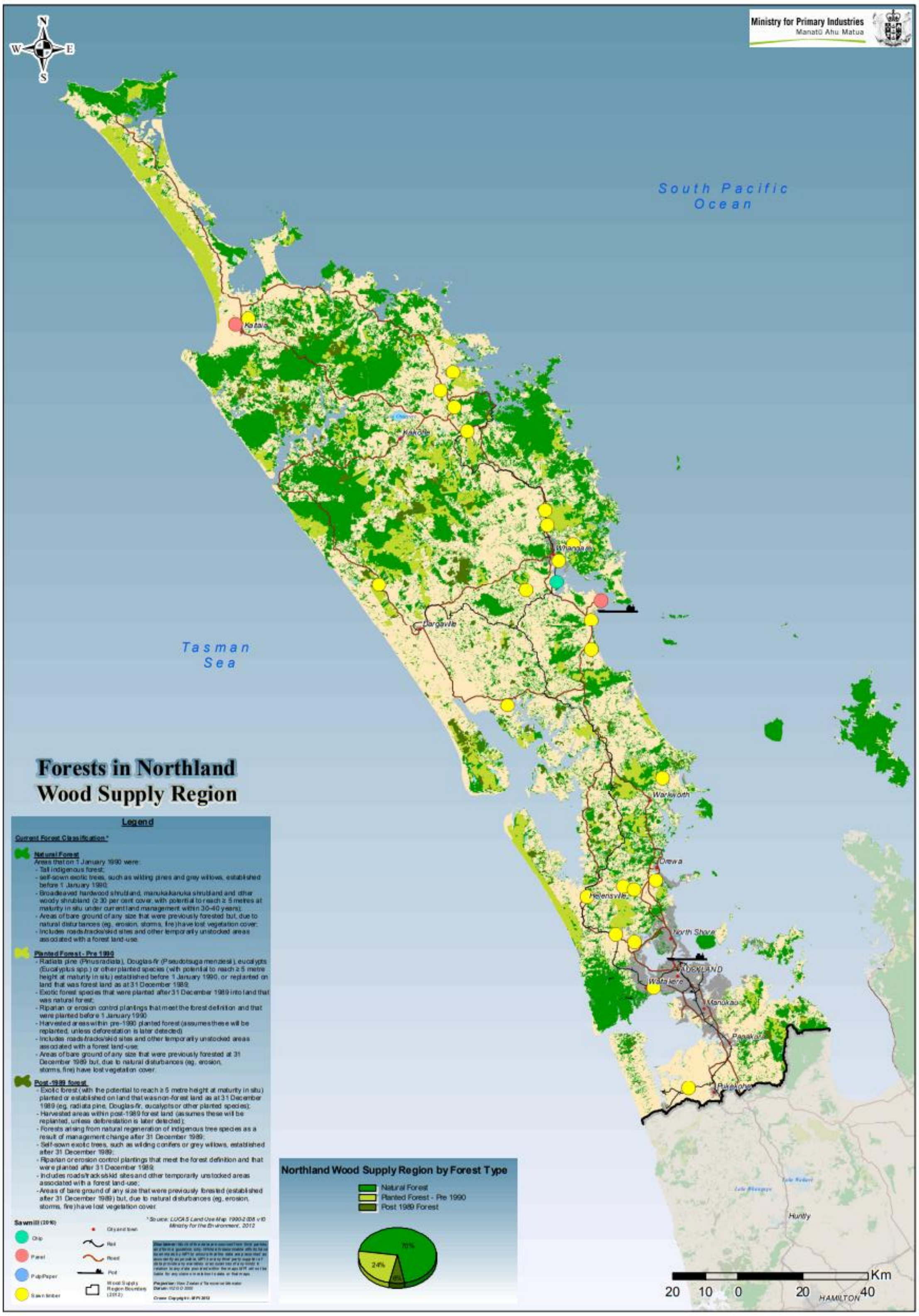

Figure 3: Wood supply region - Northland, showing natural forest, planted forest - pre 1990 and post 1990 forest (MPI, 2012)

The National Exotic Forest Description (NFED) survey provides information on forestry in NZ but is segregated into greater than 40 ha (main survey) and those less than 40 ha in the NEFD Small Forest Survey. The NFED for plantation forests provides information on plantation forestry by territorial authority as of 1 April 2021 (Table 2, Table 3 and Table 4). The 'plantation forest HRU' in the FWMT is defined as exotic forest plantations greater than 10 ha in extent.

As can be seen from these Table 4 the primary plantation forestry in Auckland is *Pinus radiata* (97%). It also indicates that there is not a consistent age band, with approximately 27% of *Pinus radiata* in the 21–25-year age band. This means that a significant proportion of radiata pine could be harvested in the next 5 years, which will have a larger impact than in 25 years when the 8% of *Pinus radiata* that is currently 1-5 years old is harvested. To better represent plantation forestry in the FWMT, a decision is required about whether to continue representing the activity as a generalised whole-cycle, or to disaggregate the recently harvested phase (and potentially other stages). Presenting a disaggregated forestry cycle into different HRU groupings would enable more accurate classification of land use activities and therefore better estimates of base footprints, mitigation opportunities and mitigation cost and effectiveness, however it would require significantly more evidence.

Table 2: Area, standing volume and area-weighted average age by territorial authority as of 1 April 2021 (MPI, 2021)

Territorial Authority	Net Stocked Area (ha)	Standing volume ('000 m^3)	Area-weighted average age (yrs)
Far North District	86,658	19,406	16.9
Whangarei District	31,309	6,843	16.5
Kaipara District	38,128	7,747	15.2
Auckland Council	37,929*	11,239	20.5
Northland region total	194,024	45,235	17.2

* *Note that this may be slightly different to the plantation forest in the FWMT (37,637 ha in Table 1) which is likely due to different reporting years and data sources.*

Table 3: Forest area by forest owner national size class, as of 1 April 2021 (MPI, 2021)

National size class (ha)	< 40	40 – 99	100 – 499	500 – 999	1,000 – 9,999	>10,000	Total
Northland wood supply region	35,556	7,032	14,609	8,930	17,105	110,791	194,023

Table 4: Area planted by forest type and age class for Auckland Council territorial authority as of 1 April 2021(MPI, 2021)

Area planted in (ha)	Age class (yrs)										
	1-5	6-10	11-15	16-20	21-25	26-30	31-35	36-40	41-50	51-60	61-80	Total
All species	2,824	2,604	4,230	8,212	10,247	7,142	470	720	1,354	96	29	37,928
Pinus radiata	2,813	2,594	4,173	8,059	9,947	6,812	389	638	1,259	85	10	36,779
Douglas-fir	0	6	0	0	0	0	0	0	0	0	0	6
Cypress	0	3	15	51	31	113	16	4	4	1	0	238
Other softwoods	C	0	3	40	42	53	8	25	16	2	3	C
Eucalypts	0	C	27	18	C	11	4	9	10	3	0	88
Other hardwoods	C	0	12	44	222	153	53	44	65	5	16	C
Pinus radiata **pruned with production thinning**	0	112	35	742	1,353	1,749	152	233	463	62	1	4,902
Pinus radiata **unpruned with production thinning**	0	0	1	33	211	450	5	0	0	0	0	700
Pinus radiata **pruned without production thinning**	99	373	C	686	2,150	1,916	109	195	692	2	2	C
Pinus radiata **unpruned without production thinning**	C	2,109	3,618	6,598	6,233	2,697	123	210	104	21	7	C

(C = confidential and data was not provided in NFED).

2 Requirements of Plantation Foresters: NES-PF

This section highlights the background and key parts of the National Environmental Standards for Plantation Forestry (NES-PF) relevant to water quality management. It is included in this report as a key consideration for the FWMT build as the counterfactual, or the baseline, from which to consider the opportunity for further mitigations and changes to forestry management and the associated performance of these mitigations.

Importantly, any baseline should consider both the NES-PF and relevant regional policy. Any baseline state (management actions) can be assessed against the monitoring and non-compliance data (where available) or in the absence of information, assuming the NES-PF, especially the permitted activity rules, are complied with.

2.1 Background to the NES-PF

The NES-PF provide regulations under the Resource Management Act 1991 (RMA) and came into force on 1 May 2018. These regulations provide nationally consistent standards that aim to maintain or improve the environmental effects of plantation forestry activities and increase the efficiency and certainty of managing such activities (MPI, 2020a). The NES-PF prevails over district or regional plan rules except where certain plan provisions may be more stringent than the NES-PF and where forestry activities fall outside of the NES-PF's definition of plantation forestry; in which case, existing district or regional plan provisions may apply. In essence, the NES-PF ensures that the majority of forestry activities across the country do not require resource consent and operate as permitted activities subject to compliance with the regulations in the NES-PF.

Plantation forestry under the NES-PF is defined as any forest deliberately established for commercial purposes that is at least one hectare in size of continuous cover forest species which will be harvested. Forest species are defined as tree species that are capable of reaching five metres in height at maturity. Plantation forestry does not include shelter belts where the average width of the tree crown cover is less than 30 metres, forest species in urban areas, nurseries and seed orchards, trees grown for fruit or nuts, long-term ecological restoration planting of forest species or willows and poplars space planted for erosion control. The intention of the NES-PF is to apply to plantation forests that have been deliberately established with an intent to harvest for commercial gain.

The NES-PF covers eight core plantation forestry activities. These are provided, along with their definitions, in Table 5. The NES-PF uses a risk-based approach to determine the consent requirements for each of these forestry activities. Plantation forestry activities are generally permitted with relevant activity conditions, where good management practice is complied with. Management plans for some activities (earthworks, forestry quarrying, and harvesting) are required and favoured over more prescriptive rules. Where permitted activity conditions cannot be complied with, or where the risk of adverse environmental effects is heightened, resource consent is required. This may be as a controlled activity, restricted discretionary activity or discretionary activity.

Table 5: Definition of the eight plantation forestry activities regulated under the NES-PF. Adapted from the NES-PF User Guide (MPI, 2018).

Forestry activity	Definition in the NES-PF	Associated regulation(s)
Afforestation	Means planting and growing plantation forestry trees on land where there is no plantation forestry and where plantation forestry harvesting has not occurred within 5 years. Does not include pre-planting vegetation clearance. This includes rules that afforestation must not occur: • Within 5 m of a perennial river with a bankfill channel width of less than 3m, or a wetland larger than 0.25 ha or, • Within 10 m of a perennial river with a bankfill channel width of greater than 3m, or a lake larger than 0.25 ha or,an outstanding freshwater body, or a water body subject to a water conservation order, or a significant natural area or, • Within 30 m of the coastal marine area.	8-17
Pruning and thinning to waste	Means pruning plantation forests trees and thinning to waste involving the selective felling of plantation forest trees within a stand where the felled trees remain on site.	18-21
Earthworks	Means disturbance of the surface of the land by movement, deposition, or removal of earth (or any other matter constituting the land, such as soil, clay, sand, or rock) in relation to plantation forestry; and • Includes the construction of forestry roads, forestry tracks, landings and river crossings approaches, cut and fill operations, maintenance and upgrade of existing earthworks, and forestry road widening and realignment; but • does not include soil disturbance by machinery passes, forestry quarrying, or mechanical land preparation	22-35 (reg. 3 – definitions for forestry roads, tracks, landings and maintenance and upgrade of existing earthworks)
River crossings	Means a structure that is required for the operation of a plantation forest and provides for vehicles or machinery to cross over a water body; and • includes an apron and other structures and materials necessary to complete a river crossing; but • does not include a stormwater culvert or a culvert under a forestry road or forestry track	36-49 (reg. 3 – definitions for types of river crossings)
Forest quarrying	Means the extraction of rock, sand or gravel for the formation of forestry roads and construction of other plantation forestry infrastructure, including landings, river crossing approaches, abutments, and forestry tracks, i. within a plantation forest; or ii. required for the operation of a plantation forest on adjacent land owned or managed by the owner of the plantation forest; and • includes the extraction of alluvial gravels outside the bed of a river, extraction of minerals from borrow pits, and the processing and stockpiling of material at the forest quarry site; but • does not include earthworks, mechanical land preparation, or gravel extraction from the bed of a water body.	50-61
Harvesting	Means felling trees, extracting trees, thinning tree stems and extraction for sale or use (production thinning), processing trees into logs, or loading logs onto trucks • Does not include – i. Milling activities or processing of timber; or ii. Vegetation clearance of vegetation that is not plantation forest trees	62-71
Mechanical land preparation	Means using machinery to prepare land for replanting trees, including root-raking, discing, ripping, roller crushing, cleaning slash, and mounding the soil into raised areas; but • Does not include –	72-75 (reg. 3 - definitions for methods referred to in the

	i. The creation of alternating drains and planting mounds using a V-shaped blade attached to the front of a bulldozer, or ii. Earthworks or forestry quarrying.	definition of mechanical land preparation)
Replanting	Means the planting and growing of plantation forestry trees on land less than five years after plantation forestry harvesting has occurred.	76-81

In addition to the NES-PF, the Auckland Unitary Plan contains definitions and associated rules for quarrying and earthworks that are ancillary to forestry (e.g., Auckland Unitary Plan, E1.6.4). These provisions apply outside the bounds of the NES-PF (i.e., where the NES-PF definitions for "plantation forestry," "earthworks" or "forest quarrying" have not been met).

The NES-PF utilises three risk assessment tools to help determine the level of environmental risk from a forestry activity. These tools include the Erosion Susceptibility Classification (ESC), the Fish Spawning Indicator and the Wilding Tree Risk Calculator.

Only the ESC is discussed here given its direct link to water quality and the objectives of the FWMT. The ESC database is a key tool used to identify land susceptible to erosion and set regulatory thresholds under the NES-PF for each of the regulated forestry activities. The ESC dataset is based on the New Zealand Land Resource Inventory (NZLRI) and classification of Land Use Capability (LUC) units. The ESC categorises land into four zones (green, yellow, orange and red) on the basis of topography, dominant erosion process and rock type which informs erosion susceptibility.

Green (low risk) and yellow (moderate risk) zoned land is less likely to erode and therefore plantation forestry activities are permitted provided the relevant conditions are complied with (e.g., good management practice). In contrast, orange (high risk) and red (very high risk) zoned land is more likely to erode. Most forestry activities require a resource consent on very high-risk land, whereas only some activities on high-risk land (e.g., earthworks where slope is above 25°) require resource consent (Table 6). Where a plantation forestry activity occurs on multiple ESC zones, including zones requiring resource consent, then the resource consent is only required for the part of the activity and land that falls within the ESC zones requiring consent (MPI, 2018).

On red zone land, a two-hectare threshold is applied to determine whether some activities are permitted, or resource consent is required. These activities include afforestation within a calendar year, harvesting within a three-month period, mechanical land preparation within a calendar year or replanting within a calendar year. For example, provided all other relevant permitted activity conditions are complied with, afforestation on red zoned land is permitted if two hectares or less is planted within a calendar year. The purpose of this rule is to allow the small extent of an activity to be permitted within a higher risk zone where resource consent would otherwise be required (MPI, 2018).

The ESC is the initial step used to screen land for forestry activities on the basis of environmental risk. However, the ESC mapping is based on a scale of 1:50,000, reflecting the resolution of the NZLRI. At 1:50,000, the minimum legible area on a paper map is 10 ha (Te Uru Rākau, 2019). It is therefore not suitable for use at the operational level. While the ESC could help inform the opportunity for mitigation options for forestry activities, the resolution of slope and soils information in the FWMT is more granular than the ESC.

Boffa Miskell (2016), assessed the effectiveness of the NES-PF provisions to manage erosion against existing provisions in a selection of RMA plans in nine regions. The assessment concluded that the NES-PF provisions for plantation forestry activities associated with erosion "work well to minimise effects" and were given the highest grading in this assessment. In comparison, the effectiveness of provisions in existing plans were often given a lower "provisions manage effects to a tolerance threshold" rating. This indicates that the provisions in the NES-PF to manage erosion generally represent a high level of practice compared to the status quo. However, it should be noted that the Auckland region was not one of the regions explicitly assessed in this comparison.

Table 6: Consent requirements based on Erosion Susceptibility Classification (ESC) zone. Adapted from the NES-PF User Guide (4Sight Consulting Limited, 2018).

ESC Zone	Green	Yellow	Orange	Red
Risk rating	Low risk	Moderate risk	High risk	Very high risk

Threshold for consent requirement	n/a	n/a	• **Earthworks** on a slope of 25 degrees or more and, in any 3-month period, comprise:
			  • side cutting over a height of 3 m or side cutting of a continuous length over 100 m; and
			  • the deposition of more than 500 m^3 of spoil or fill.
			• **Forestry quarrying** in earthflow terrain.
			• **Mechanical land preparation** where the land slope is 25 degrees or more, the subsoil is affected, and the area covered by the mechanical land preparation activity is greater than 2 ha in a calendar year.
				• **Afforestation** where the land proposed for afforestation is greater than 2 ha in any calendar year.
				• **Earthworks** in any 3-month period which comprise
				  • side cutting over 2 m or side cutting of a continuous length over 50 m; and
				  • deposition of more than 100 m^3 of spoil or fill.
				• **Harvesting** where the land is Land Use Capability Class 8e and it involves more than 2 ha of harvesting in any 3-month period.
				• **Mechanical land preparation** where the land slope is 25 degrees or more, the subsoil is affected, and the area covered by the mechanical land preparation activity is greater than 2 ha in a calendar year.
				• **Replanting** where the land proposed for replanting is greater than 2 ha in any calendar year.

2.2 Potential impact of forestry activities in the NES-PF to water

Discussion of the individual forestry activities and potential impacts to the environment are summarised in Table 7 below.

Table 7: The potential impact of the eight plantation forestry activities regulated under the NES-PF to water quality and contaminants involved. Adapted from the NES-PF User Guide (MPI, 2018).

Forestry Activity	Potential Impact on Water Quality	Contaminants Involved
Afforestation	• Generally related to the longer-term effects associated with the location where the forest is established rather than from the planting itself. • Soil slip erosion. • Afforestation needs to also consider the effects of future forestry activities on the receiving environment.	Sediment Phosphorus
Pruning and thinning to waste	• Deposition of slash to water.	Predominantly impacts habitat
Earthworks	• Earthworks have the potential to cause contaminant loss to waterways where works are not properly planned and managed. • Generally, first rotation forests pose a greater risk of adverse environmental outcomes as there is a greater amount of earthworks to be completed. • Accelerated erosion. • Excessive sediment discharge to water.	Sediment Phosphorus
River crossings	• Sedimentation of the river during construction and use.	Sediment Phosphorus
Forest quarrying	• Forestry quarrying has similar potential for adverse environmental impacts relating to soil and slope stability and sedimentation. • Sedimentation of water bodies.	Sediment Phosphorus
Harvesting	• Soil disturbance from harvesting. • Post-harvest soil erosion as tree roots and stumps rot. • Deposition of slash to water.	Sediment Phosphorus
Mechanical land preparation	• Sedimentation discharge to waterways. • Activation of erosion-prone areas.	Sediment Phosphorus
Replanting	• Similar adverse effects to afforestation are possible following replanting, however, are generally less significant as forestry infrastructure is already in place and therefore soil disturbance from upgrades and maintenance will be less than the initial construction.	Sediment Phosphorus

2.3 Ancillary activities

In addition to the eight core plantation forestry activities, the NES-PF also regulates three other ancillary activities. These are:

- Slash traps
  A slash trap is defined in the NES-PF as a structure set in a river, on the bed of a river, or on land to trap slash mobilised by water. Slash is defined as the tree waste left behind after plantation forestry activities. If not well designed and managed, slash traps have the potential to cause adverse environmental effects. If debris builds up behind a slash trap and is not removed, this can cause scouring of the riverbank or bed as the water attempts to move past the structure.

- Indigenous vegetation clearance
  Indigenous vegetation in the NES-PF is defined as vegetation that is predominantly vegetation that occurs naturally in New Zealand or that arrived in New Zealand without human assistance. Vegetation clearance is defined as the disturbance, cutting, burning, clearing, damaging, destruction, or removal of vegetation that is not a plantation forest tree. It does not include any activity undertaken in relation to a plantation forest tree. Impacts to water quality from indigenous vegetation clearance are around soil disturbance and erosion.

- Non-indigenous vegetation clearance
  There is no specific NES-PF definition for non-indigenous vegetation. Any vegetation that does not meet the NES-PF definition of *indigenous vegetation* is generally accepted to be *non-indigenous vegetation*. The same vegetation clearance definition and impact are therefore as described for indigenous vegetation clearance above.

2.4 New Zealand Environmental Code of Practice for Plantation Forestry

In addition to the NES-PF there is a New Zealand Environmental Code of Practice for Plantation Forestry (the Code; NZFOA, 2015). The Code aims to be a key reference tool for those managing forests by providing information on environmental values, how such values should be assimilated into operational planning, other references and resources as well as the best environmental practices (BEPs). In this sense, it is akin to the GMP guidelines used in the pastoral sector—the Matrix of Good Management Practice[1].

The BEP section is intended as a tool-kit and describes a range of management options that could be applied to a particular situation. It is up to the forester to consider all the relevant factors, on a site-by-site basis, and decide which one (or more) of the options provided is most applicable. The provision and exercise of such discretion for the forester means that it is not suitable or appropriate for any regulatory body to require blanket compliance with this code (NZFOA, 2015).

The Code is voluntary and it highlights a range of actions and 'rules' and 'guidelines' associated with each action. The rules in this case relate to the actions needed to meet the Code's requirements of BEP. The Code also highlights potential adverse impacts of each of the activities in the BEPs. However, because of the objectives and structure of the Code there is no quantification of costs and benefits associated with each of the BEPs. Based on the recommendations in the Code, forest operators select

[1] https://www.ecan.govt.nz/document/download?uri=2378592

which BEPs are most suited and applicable to their situation and thus are adopted. A summary of the BEPs can be found in Appendix 1.

2.5 Implications for the FWMT

The use of permitted activity conditions (including management plans in some cases) and the resource consent pathway within the NES-PF was the focus of the Section 32 analysis when the NES-PF was proposed. This means that the Section 32 and regulatory impact statements focus on the cost of management plans and/or obtaining resource consent, not the costs and benefits of alternative practices in forestry. This is largely due to a significant lack of information available to inform cost–benefit analysis on forestry management practices (i.e., on anything more than a generalised set of practices equivalent across all plantation forest types). However, an overview of the NES-PF is provided here because it is a consideration for the FWMT build as it provides the baseline from which to consider further mitigations and changes to forestry management, the associated implications of these mitigations as well as the opportunity for mitigation.

The permitted activity and management plan process which forms the basis of the regulatory approach within the NES-PF is similar to the industry-led GMP process for farming and horticulture activities. The management plans required under the NES-PF permitted activity classification focus on showing how forestry activities will meet the NES-PF. The actions completed under a permitted activity status are particularly hard to estimate as there is no readily accessible register of these actions. Within the permitted activity bounds there is variation in how foresters implement actions and therefore there is limited cost/benefit or opportunity information for the various actions that could be used to inform mitigation assessment. Given the FWMT considers additional costs of management beyond business-as-usual activity, the lack of verifiable information on permitted activity management and the likelihood any impacts of permitted activities being adopted are minor (as voluntary), the latter effects might need to be assumed already expressed in baseline FWMT results (i.e., dynamic intervention modelling can only assume permitted activity GMP is already resulting in effects on baseline water quality). Alternatively, the NES-PF could be applied as the initial mitigation bundle so a sensitivity analysis can be run on the NES-PF being fully adopted in the base state, or not. Hence, baseline outputs from the FWMT can be assumed to already incorporate actions required from the NES-PF. Doing so, results in similar assumptions to pastoral and horticultural intervention modelling in the FWMT Stage 1, where baseline projections are assumed to already incorporate the effects of voluntary industry action, even if the exact extant of these can't be perfectly quantified.

The NES-PF utilises three tools to assess environmental risk. The ESC categorises land on the basis of topography, dominant erosion process and rock type which informs a generalised erosion susceptibility. This is not explicitly represented by the FWMT HRU typology. However, areas of ESC categories could be explicitly or implicitly used to disaggregate Plantation Forest into sub-types for the purposes of defining source control opportunities (including associated costs and effects). The ESC could, in theory, be used to help define the opportunity for mitigations to be applied to plantation forestry land. However this would require a level of data that is not currently available.

Whilst forestry management decisions are made at a finer scale than the ESC maps (1:50,000), this presents little challenge to using ESC information to determine source control opportunities in the FWMT, as the latter optimises to sub-catchment scale (ca. 40-100 ha).

However, information is lacking to develop unique LCC or benefit estimates for each ESC or HRU types. In the immediate future, generalisation of expected cost and benefit to contaminant yield is necessary.

The lack of information on the costs and benefits of the action's foresters take under the NES-PF or the Code means that it is challenging to incorporate some or all of these actions as mitigations in the FWMT. The NES-PF should be considered as the counterfactual, and it should be assumed that all Plantation Forestry is compliant with the relevant requirements. The Code is voluntary and often over and above the permitted activity requirements in the NES-PF and so it does not make sense for this to be a part of the counterfactual. Actions in the Code can be considered as mitigations where there is information on the costs and benefits.

2.6 Summary

This section has highlighted the current policy requirements and industry recommended practices for plantation forestry. This section helps define the opportunity for device and source control practices across forestry in the FWMT as well as contributing to the understanding of forestry's base footprint.

The NES-PF could form the basis for defining the starting point or 'base state' in the FWMT from which mitigation is applied. Alternatively, it could form the basis for the first mitigation bundle. In this sense the NES-PF would form the basis for mitigation bundle 1 (or as a proxy for good management practice). From this point the Code would be considered a further mitigation bundle or best management practice. More information is required to estimate the current extent of practices across forestry in the Auckland region to help define the current practices, the potential for further mitigations to be applied and, accordingly, the appropriate approach for incorporating the NES-PF and the Code into the FWMT.

One key point is that the NES-PF has requirements for 5 m riparian buffers. If it is assumed the NES-PF is fully adopted in the base scenario then there is no more opportunity for 5 m buffers, only for buffers greater than 5 m. If it is assumed that the NES-PF is not yet adopted, then 5 m buffers could be adopted everywhere. While it is not clear how many forestry areas actually have 5 m riparian buffers around waterways, it is assumed that these are widely used across the forestry sector. This means that there is opportunity for wider riparian areas to be incorporated.

3 Base State Profit and Water Quality Impacts

3.1 Introduction

This section reviews the literature on the base state of forestry, including both the economic and environmental performance. Forestry can be used for a range of uses including, timber, carbon credits, and mānuka honey production, as well as a permanent offset for water quality impacts. Plantation forestry in the Auckland region, as previously discussed, is most likely to be the species *Pinus radiata*.

Given the long-time horizons to realise income from plantation forestry (approximately 27-years for *Pinus radiata*) and the variations in costs and benefits over time, estimations of costs and benefits are often best done in a cost–benefit framework. The LCC framework used by the FWMT is based on a 50-year time frame which lends itself to this long-term nature but does not readily align with the timeframes for full rotations of plantation forestry. This is important given the impact on water quality varies markedly at different times in the rotation period (e.g., at planting, maturity, and harvesting).

Pinus radiata can have a difference in harvest age depending on site and timber use. For typical use in timber framing, a rotation length of approximately 24-30 years is required (NZFFA, 2021). However, trees tend to need to be at least 25 years old to produce high quality timber. It would be most realistic to assume a rotation cycle is at least 27 years with harvest and replanting time, meaning a 56-year period would see two full rotations. This represents a challenge for the FWMT which is using a 50-year LCC framework.

The long-term nature of forestry means researchers and industry tend to utilise metrics such as the internal rate of return (IRR). The IRR is the discount rate that makes the net present value (NPV) of all cash flows associated with an investment equal to zero in a discounted cash flow analysis. Consequently, the IRR is not directly comparable to metrics for operating profit used in horticultural and pastoral businesses. An equivalent annual annuity (EAA) calculates the constant annual cash flow generated by a project over its lifespan as if it was an annuity. Using the 50-year LCC framework, and ultimately NPV, in the FWMT is similar to the use of IRR.

As with other rural land uses, the profitability and water quality impacts of forestry vary across forest businesses and location. Factors affecting water quality impact include biophysical context (e.g., slope, soil, geology, climate, pests). Factors affecting profit include rate of growth (climate, management), location (distance to port) and cost of harvest. This section reviews the information available on the base state of forestry businesses, both economic and environmental.

There has been a limited pool of data available on forestry profitability and water quality impact (Baillie and Neary, 2015). The long-term nature of forestry coupled with varying financial and environmental impacts at rotational stages, compounds the lack of relevant datasets. The lack of data presumably reflects the use of plantation forest as a land use change option to manage to a limit-based approach under the National Policy Statement for Freshwater Management (NPS-FM), with limited data on water quality performance under differing plantation forestry scenarios.

NZIER (2016) completed three cost–benefit analyses (CBAs) of the NES-PF for MPI. The first two CBAs (2011 and 2012) suggested that the costs of the NES-PF outweighed the benefits, while the 2014 analysis indicated that the benefits may marginally outweigh the costs. The differences in the 2014 results relative to 2011 and 2012 CBAs resulted from:

- Changes in the operating environment (e.g., changes to the Climate Change Response Act),
- Further refinement of data relating to stream setbacks.

- Changes within the status quo due to regional and district plan amendments and also the National Policy Statement for Freshwater Management.
- The removal of quantified environmental benefits associated with sediment loss based on discussions with councils and forestry managers to reflect the lack of knowledge of the quantifiable environmental benefits associated with moving from a five metre to a ten-metre setback NZIER (2016).

While the CBAs for National Environmental Standards often quantify costs and benefits associated with actions designed to improve environmental outcomes, the CBA completed by NZIER (2016) on the NES-PF focuses on the costs to forestry companies and councils from complying with the changed policy requirements (administration costs not management changes in forests) and benefits (mainly a reduction in plan costs), rather than the costs and benefits associated with changing practices in the forests (with the exclusion of considering setbacks). The environmental benefits that are assessed are largely qualitative not quantitative.

NZIER (2016) used interviews and a literature review and found little concrete evidence to support further quantifying environmental benefits. For example, the literature reviewed by NZIER (2016) suggested for example, that increased setbacks will have an environmental benefit. However, the NES-PF steering group decided that there was insufficient evidence to quantify a marginal change in waterway setback. In earlier CBAs, NZIER (2011; 2012) attempted to quantify environmental benefits using values generated by Krausse et al (2001) and the Landcare NZEEM model. This approach was criticised by the NES-PF forestry working group due to the lack of robust quantifiable estimates and was not integrated in the 2014 CBA. In addition, the Section 32 Analysis for the NES-PF noted that "this assessment does not quantify environmental benefits and costs as there are some inherent difficulties and large uncertainties" (MPI, 2017). Similarly, the environmental impact assessment of the NES-PF undertaken by Scion (2015a) did attempt to quantify some environmental benefits and costs but noted that the values provided are indicative rather than definite and only provide an order of magnitude of the potential benefits."

3.2 Profit

The profitability of plantation forestry depends on factors such as tree species, growth rates, establishment and maintenance costs, grades of the logs, pruning regime, timber prices and markets, climate and location (Jones et al. 2008). The log market is vulnerable to global economic fluctuations and there is a long time between planting forestry stands and harvest making it very difficult to predict the state of the log market at harvest time. Profit will also depend on what is included as 'product' and if it includes only timber, or if the returns are calculated to include carbon, the assumptions around the timing of carbon value realisation, and the liability if carbon is claimed and then the forest is harvested.

Where possible this analysis separates out the two components and focuses primarily on timber returns. However, the majority of the analysis that includes carbon does so based on the recently applicable averaging approach for farm and plantation forest types which means forests return carbon for approximately 16 years with no liability to pay the credit back at harvest provided the land is replanted; for permanent forests, carbon revenue is included as carbon is generated as long as carbon is sequestered, which is at least 50 years based on the MPI Look Up Tables.

It can be challenging to model forestry as the length of the investment period can make it difficult to estimate potential costs and returns over the investment lifecycle. Fundamental shifts in pricing, can occur, for both input costs such as harvesting and for product prices, by the time the investment is realised. Returns are also influenced by central government climate change policy (e.g., afforestation grants and carbon regulations) as well as regional or central government sustainable land management grants.

The returns from forestry will depend if the estimate is for on-farm forestry or commercial plantation forestry. There is, however, often no clear-cut distinction between the groups. It is often assumed that on farm plantation forestry is typically smaller in size (ca. 40 ha or less), however, this can include small scale commercial forestry lots as well (as seen by the FWMT using 10 ha as the lower limit for plantation forestry).

There are typically four profit measures that can be used in analysing forestry returns: net present value (NPV), land expectation value (LEV), equal annual equivalent (EAE; another name for EAA), and internal rate of return (IRR) (Olssen et al., 2012). NPV and IRR are the most common. The NPV of an investment is the sum of the present value for each year's net cash flow less the initial cost of the investment. Investments with a positive NPV mean that the investment generates a return greater than the assumed discount rate; those with a negative NPV generate a lower return than the assumed discount rate and would be rejected. The IRR is the actual rate of return on an investment accounting for the time value of money – essentially the discount rate at which the NPV of an investment would be zero. Long-term cashflows are sometimes considered using an annuity (an annual cashflow value that would deliver the same NPV over the lifetime of the investment at the assumed discount rate as the investment itself) to compare forestry investments with alternative land uses.

Challenges when comparing financial returns from forestry with other land uses, include the use of different metrics, the treatment of land opportunity costs and the length of time considered which interact with the choice of discount rate. Some assessments of profitability across pastoral and forestry land uses exclude land value on the basis that land is a sunk investment for all of the scenarios considered (e.g., Dooley & Muller, 2021). Others suggest that because the returns from farming are not always differentiated from the returns due to appreciating land values, analysis should only consider the returns from production as a measure of comparison (e.g., Evison, 2008a).

There has been a limited pool of data available on forestry profitability. Up to the mid-2000s there was no publicly-available, regularly-produced estimates of the profitability of forestry (Evison, 2008a). However, now there is more data being published in the Forestry Facts and Figures annual publication (published by the Forest Owners Association annually). There are also more studies being published which include profitability estimates and models being used to estimate profitability under a number of scenarios.

Olssen et al. (2012) estimated forestry profitability in 2008 using NPV, LEV, EAA and IRR (though only NPV and IRR are presented here) by region, selected regions are presented in **Error! Not a valid bookmark self-reference.**. Profitability was based on the MPI data for large forest owners and may not be representative of the price that can be obtained by small forest owners. However as described in Olssen et al. (2012), typically a few forest owners have held most of country's forest land (70% of plantation forestry land in 2010 was owned by large[2] forest owners). To estimate profit, they estimated revenue for unpruned logs by aggregating the price information on different grades of unpruned logs and utilised an 8 % real discount rate. Profits were larger for regions that are further north due to better wood yield as well as typically lower logging and cartage costs. The estimates in this study excluded carbon revenue.

[2] Where large forests are greater than 40 ha in line with the National Exotic Forest Description

Table 8: NPV, IRR and EAAfor *Pinus radiata* in selected regions 2008, adapted from Olssen et al. (2012)

Wood supply region	NPV ($)	IRR (%)	EAA ($/yr)
Auckland	1,337	9.90	121
Northland	1,180	9.64	107
Central North Island	1,135	9.62	103
NZ weighted average	647	8.75	59

Praat and Handforth (2019) estimated earnings before interest and tax (EBIT/ha) as an annuity of $292 and 7.9% IRR for a farm plantation forest that is solely harvested for timber, and $744/ha and 14.7% IRR for an on-farm plantation forest utilised for timber & carbon (at $25/t carbon). Evison (2013) used surveys of commercial forestry costs in Nelson/Marlborough and Otago/Southland combined with prices and yields published by MPI, indicated commercial forestry was providing IRRs of 5.8% and 5.6% respectively (Evison, 2013). The IRR in Evison (2013) is lower than Praat and Handforth (2019); however, Evison (2013) is based on commercial forestry rather than farm plantation forestry and is for South Island regions, not national averages as in Praat and Handforth (2019). The IRR in Olssen et al. (2012) is higher for both national and Auckland than Praat and Handforth (2019) and Evison (2013).

Neilson (2010) summarised typical IRR for *Pinus radiata* since the mid-1990s. The IRR was estimated to be 4% and 4.4% for structural sawlog and pruned sawlog regimes respectively, in 2009. Meanwhile, Liley (2010) analysed the IRR for a large number of forestry investments in New Zealand and estimated an average of 5.8%, with a range from 3.2 to 8.2%. This is similar to the estimates of IRR collected in a survey carried out by Manley (2010) which ranged from 1 to 7% pre-tax. These are all slightly lower than that in Evison (2013), Praat and Handforth (2019) and Olssen et al. (2012).

The 2019-2020 Facts and Figures (NZFOA, 2020) provides an annual cash surplus utilising a 10-year average. Forestry is assessed at approximately $1,200/ha. The forestry result appears to relate to a national average covering all forestry, though this is not clear. While this is based on 2019-20 data and Olssen et al. (2012) is 2012 data this is significantly higher than the national weighted average presented in Olssen et al. (2012), though close to data presented for northern regions such as Northland and Central North Island.

Yao et al. (2021) notes that timber production from existing planted forests varies in profitability with annualized NPVs ranging between -$200 and $900 per ha per year based on Forest Investment Framework's (FIF's) spatially explicit timber profitability analysis. The average profit per hectare for a timber only scenario in Auckland was $405/ha/yr, utilising a 6% discount rate and a 28-year rotation compared to a national average of $314/ha/yr. This is significantly lower than some of the NPVs presented in other studies such as Olssen et al. (2012).

Evison (2008a) derived their forestry costs from industry sources, for a typical pruned log regime and timber revenues only were included, with price information derived from MPI, the same method as Evison (2013). All IRRs are real, pre-tax. The model assumed an investment period of 30 years and constant real costs and returns over the entire investment period. They estimated that for a 500 ha (effective) forest the IRR was 2.71%, with a cash surplus of $77,193 and a capital value of $5,700/ha. They also indicated net cash income ($682,023), working expenses ($554,825) and management costs ($50,006) for the forestry model (Evison, 2008a).

Daigneault, Wright and Samarasinghe (2015) estimated an annual mean net return of $680/ha for plantation forestry in the Maniapoto area. This was based on assumptions presented in Hock et al. (2014), who used the Forest Investment Finder (see Section 3.2.1) to estimate the total NPV of the plantation over a 28-year rotation. Carbon was included (at a value of $4/NZU) and the NPV was annualised by multiplying the figure by an equal annualised equivalent factor of 0.06.

Evison (2010) used a similar approach to Evison (2008a). Evison (2010) estimated the annual profitability (including separating out some cost categories) of various land uses in 2010 based on a discounted cash flow model for each land use option. The purpose was comparing the profitability across land uses and as such the IRR is the value of focus. They estimated that for a 500 ha (effective) forest the IRR was 2.32%, with a cash surplus of $61,223 and a capital value of $2.644 million. They also indicated net cash income ($754,983), working expenses ($638,831) and management costs ($54,929) for the forestry model.

Park (2011) assessed the profitability of small-scale forests in the Whanganui District. In a valuation of existing small-scale forests that only produce timber they found an average NPV (not annualised) of $575/ha with a range between $-2,804/ha and $6,852/ha at optimum harvesting age. The average optimum rotation age was 35. There were also 27 (out of 58) forests in the sample that were eligible to enter the ETS. The implementation of the ETS (at a carbon price of $20/NZU, not using a carbon averaging approach) results in an increase in both rotation age and profitability of the existing forests (Park, 2011). The average optimum rotation age shifts to 49, and all forests have positive NPV with an average value of $5,609/ha. When considering the sample forests as new plantings, the profitability as represented by IRR of production forestry based on the estimated costs ranges from 1.6 to 4.7% (2.8% on average) along with five forests with negative profitability due to extremely high production costs (Park, 2011).

Matheson et al., (2018) used a discounted cashflow analysis (utilising a discount rate of 5%) to estimate profitability of the forestry land. Two forestry models were considered – one a *Pinus radiata* plantation model and the other a mānuka plantation established for honey production. The *Pinus radiata* model is based on a 28-year non-pruned (framing) rotation under contract management. The base model excludes the financial impact of carbon. Net stumpage at $43,490/ha and an establishment cost of $1,500/ha delivers an NPV of $6,827/ha (excluding land costs) at a discount rate of 5%. The inclusion of the value of the sale of permanently sequestered carbon (at a price of $21 /t CO_2e) in the modelling increases the NPV of this model to $9,420/ha (excluding land costs). Mānuka provided an NPV of $130/ha/year (Matheson, et al., 2018).

Perrin Ag Consultants led a series of case studies which explored integrating dairy and hill country farming with farm forestry for profitable and sustainable land use. Some of these case studies are applicable for assessing the economic impact of the farm plantation forestry HRU within the FWMT. The ones that are applicable included scenarios that were predominantly *Pinus radiata*, used a 6% discount rate and two rotations (across a 56-year period). The analysis in each case study included present values (based on forestry only section of the farm enterprise) and an annuity value. These are summarised in Table 9.

Table 9: Selected data from Perrin Ag case studies in the Integrating dairy and hill country farming with forestry for profitable and sustainable land use project, adapted from Durie et al. (2021), Durie et al. (2020), Dooley & Dowling (2021), Dooley et al. (2021), Parker & Dowling (2021a; 2021b), and Parker et al. (2020).

Scenario	Present value ($/ha)	Annuity ($/ha)	Present value ($/ha)	Annuity ($/ha)
	No carbon		Incorporating carbon	
Case study 1: Holdem Farm (2017) Ltd.				
17.7 ha of *Pinus radiata* across 6 blocks	2,625	164	5,196	324
49.3 ha of *Pinus radiata* across 9 blocks	3,104	194	5,779	361
Case study 2: Taumata Farming Partnership Ltd.				
27.9 ha of marginal steep land in *Pinus radiata* woodlots (20.5 ha) and native plantings	910	57	3,328	208
27.9 ha of marginal steep land in *Pinus radiata* woodlots (15.1 ha), eucalyptus woodlots (5.4 ha) and native plantings	-274	-17	2,645	165
Case study 3: Kapenga M Trust.				
155 ha of planting, 144ha of *Pinus radiata* and 11 with manuka. Staggered planting across 4 yrs	-114	-7	2,615	162
155 ha of planting, 144ha of *Pinus radiata* and 11 with manuka. All planted at once	-86	-5	2,856	178
155 ha of planting, 144ha of *Pinus radiata* and 11 with manuka. Staggered planting across 4 yrs using modelled growth rates	762	48	3,522	220
Case study 4: MW & FK Linton.				
36.6 ha of *Pinus radiata* planting	2,623	164	5,230	326
36.6 ha of *Pinus radiata* planting with additional costs for control around forest edges	2,305	144	4,905	306
Case study 6: Verry Farming Ltd.				
Planting 186.6 ha in *Pinus radiata* assuming grants	2,605	163	5,556	347
Planting 186.6 ha in *Pinus radiata* assuming no grants and assumes an additional six years of safe carbon	1,233	77	5,407	338
Planting 181.6 ha in *Pinus radiata* and 5 ha in native plantings assuming grants available	2,618	164	5,651	353
Planting 186.6 ha in *Pinus radiata* assuming grants and staggered planting	2,811	309	5,747	633
Case study 7: Mangaweka Asparagus.				
29.4 ha of planting, 22.6 ha of structural timber (*Pinus radiata*) and 6.9 ha of low productivity specialty timber (*Pinus radiata*)	-8,307	-518	-2,426	-151
29.4 ha of planting, 22.6 ha of structural timber (*Pinus radiata*) and 6.9 ha of low productivity specialty timber (eucalyptus)	-6,064	-378	-7,655	-16
Case study 8: Lazy G Ranch.				
25.8 ha of *Pinus radiata* planting	-2,126	133	-2,298	-143

Moore et al. (2015) reviewed factors that affect the profitability of forestry and three case study assessments of farm plantation forestry in central North Island. Table 10 summarises the NPV and annuity results for the *Pinus radiata* scenarios, 28-year rotation, an 8% discount rate and a \$6/t CO_2 or \$15/t CO_2 carbon price.

Table 10: Selected data from the *Pinus radiata* farm plantation forestry scenarios, 28-year *Pinus radiata*, adapted from Moore et al. (2015)

Scenario	Minimum annuity (\$/ha)	Maximum annuity (\$/ha)	Minimum annuity (\$/ha)	Maximum annuity (\$/ha)	Minimum annuity (\$/ha)	Maximum annuity (\$/ha)
	No carbon		Carbon (\$6/t CO_2)		Carbon (\$15/t CO_2)	
Case study 1	-163	27	31	215	291	501
Case study 2	31	218	197	408	441	693
Case study 3	51	233	208	413	444	683

NZFFA (2019) provided a report of the harvest costs and returns for small scale growers (the largest forest was approximately 86 ha). Table 11, Table 12 and Table 13 summarise some of the results from this study. Not all data points are complete, however, the data set is comprehensive and useful for assessing the returns for small scale forests (the data has an average size of 15 ha). The data is provided in MS Excel and therefore can be interrogated in a variety of ways. The key values are \$30,063/ha as an adjusted net returns using June 2017 to May 2019 log prices and an adjusted nominal net annual returns \$1,156/ha/yr (NZFFA, 2019).

Table 11: Summary data for small scale farm foresters, adapted from (NZFFA, 2019)

Variable (regions are all averages)	Age at harvest (years)	Area (ha)	Final Stocking (stems/ha)	Net return per tonne (\$/t)	Net returns (\$/ha)	Adjusted Net returns (\$/ha) using June 17 to May 19 log prices	Adjusted Nominal Net annual returns (\$/yr/ha)
Average	27	15	320	43.3	25,528	30,063	1,156
Sample	111	215	84	183	215	215	110
Minimum	14	0.3	85	4.6	1,485	2,015	56
Maximum	44	86.3	650	104.1	76,293	98,419	2,941

Table 12: Average data for small scale farm foresters in selected regions, adapted from (NZFFA, 2019)

Variable	Sample size	Age at harvest (years)	Area (ha)	Final Stocking (stems/ha)	Km to Port	Adjusted Net returns (\$/ha) using June 17 to May 19 log prices	Adjusted Nominal Net annual returns (\$/yr/ha)
Central NI	59	26	24	350	147	34,173	1,263
East Coast	3	25	7	248	40	45,404	1,827
Hawkes Bay	87	27	14	425		28,133	1,662
Northland	35	28	5	280	80	30,198	1,106
Southern NI	7	26	23	383	91	25,420	996

Table 13: Selected data for small scale farm foresters, adapted from (NZFFA, 2019)

Variable	Sample size	Age at harvest (years)	Area (ha)	Ground slope	Adjusted Net returns ($/ha) using June 17 to May 19 log prices	Adjusted Nominal Net annual returns ($/yr/ha)
All	214	27	15	All	30,063	1,156
North Island	184	27	15	All	30,744	1,230
Northland	35	28	5	All	30,198	1,106
Northland	4	27	3	Flat	21,256	796
Northland	16	26	4	Flat to medium & Medium	32,424	1,236
Northland	15	30	6	Medium to steep & Steep	30,209	1,050
Northland (<10 ha)	32	28	3	All	30,790	1,119
Northland (>10 ha)	3	26	21	All	23,880	964
North Island (<10 ha)	97	27	4	All	31,314	1,308
North Island (>10 ha)	87	26	27	All	30,108	1,114

Evison (2008b) used a simple discounted cashflow analysis (single hectare, single rotation) to investigate the effect of ETS on profitability of *Pinus radiata* forestry and the implication of a change in carbon price during the investment cycle. They used a carbon price of $30/NZU and assumed that the forest owner exchanges NZUs for cash as they are earned, generating additional revenue throughout the rotation, and that all NZUs are repaid at time of harvest. The analysis generated a positive NPV of $6,422/ha under a series of assumptions; a fixed rotation age of 30 years, 7% discount rate and under a modelled carbon sequestration profile under a direct sawlog regime (using the Radiata Pine Calculator) (Evison, 2008b). However, despite the positive impact that the ETS can have on revenue for forests, others (Evison, 2017) suggest that the NZ ETS is unlikely to contribute a long-term positive impact on profitability of commercial forestry

Harrison and Bruce (2019) modelled the impact of afforestation on sheep and beef land in Wairoa using a base 1,000 ha model across 60 years (two 30-year clear wood regimes rotations) and a 5% discount rate. The two full rotations also allow the first rotation to accumulate carbon credits that can be traded, while the second rotation accumulates no tradeable carbon. Using local data, the forest was shown to produce 720 t of timber/ha at harvest. This equated to 400 stems per ha at 1.8 t per stem, with two rotations whereby harvesting occurred at year 30 and 60. Carbon credits were shown to be generated up until year 18 of the first rotation resulting in 556.7 t of tradable carbon (with no obligation to repay carbon credits provided the forest is replanted) with returns completed at five-year intervals (year 5, 10, 15, and 20) If the forest was not harvested, carbon credits were shown to generate 1079 t of tradeable carbon (no liability attached) with returns completed at five-year intervals (through to year 60). Table 14 summarises the key economic costs and incomes used in Harrison and Bruce (2019).

Table 14: Selected economic data for forestry regimes, adapted from Harrison and Bruce (2019)

Measures (all values are discounted)	Harvested – no carbon ($/ha)	Harvested – with carbon ($/ha)	Carbon farming – no harvest ($/ha)
Planting costs ($)	1,843	1,843	786
Rates, insurance and admin ($)	1,217	1,217	1,217
Thinning ($)	1,004	1,004	0
Pruning x 3 ($)	2,368	2,368	0
Discounted forest expenses	6,432	6,432	2,003
Harvest revenue ($/ha)	92,700	92,700	0
Harvest expenses ($/ha)	67,586	67,586	0
Net harvest revenue ($/ha)	25,114	25,114	0
Discounted harvest revenue ($)	7,091	7,091	0
Discounted carbon revenue ($)	0	7,751	11,389
NPV ($)	659	8,410	9,386

3.2.1 Spatial economic frameworks for forestry

There are a range of frameworks available which can be used to assess various aspects of forestry economics. Yao et al (2019) provides a summary of key frameworks, the New Zealand developed models are described in Table 15.

Table 15: Key properties of selected New Zealand based spatial economic frameworks for forests, adapted from Yao et al. (2019).

Framework	Function	Output	Aspects assessed	Scale
Forest Investment Framework (FIF) - SCION	Forest investment, estimating market and non-market values	Market and non-market values	Timber, carbon sequestration, erosion	Catchment to national
Forecaster - SCION	Forest planning, productivity prediction, profit analysis, wood quality, nutrient supply and demand	Multiple forestry outputs including yield tables per log grade, financial metrics, wood quality	Timber volume by log grade, understorey cropping and carbon sequestration	Forest plot level
Forest-Orientated Linear Programming Interpreter - Integral (originally developed by SCION)	Optimises what-if scenarios for forest and land use planning	Forest product quantities flows and stocks of resources	Timber volume, non-timber products, carbon stock, habitat, nutrient leaching, water yield, erosion	Landscape/ forest estate, region

The Forest Investment Framework (FIF) is a commercial tool held by SCION (previously called the Forest Investment Finder) and is designed to enable decision-makers to identify sub-catchments or regions across New Zealand and assess their viability of purchasing land and its conversion to forestry (Yao et al., 2019). It supports policy decisions and strategic planning on the economics of forestry scenarios and the value of ecosystem services provided by future and existing forests (natural or planted).

The FIF is a spatial economic framework for mapping, assessing and quantifying multiple forest values including timber, carbon and ecosystem services. Its timber viability component combines forest productivity surfaces with infrastructure networks, economic data (e.g., establishment, silvicultural, harvesting costs, log prices) and impedance layers (Yao et al., 2019). FIF's timber viability component is validated using data from seven case-study forests in New Zealand and results of the validation exercise suggest that FIF is a very good viability assessment tool for large scales forests (above 1,000 ha) (Yao et al., 2019) but more validation is needed for smaller scale forests.

FIF was developed for the purpose of regional and catchment analysis (Yao et al, 2016). However, at the individual forest level, it has limitations. For example, the model disregards property boundaries and differences in ages between trees in a common landscape (Hock, Harrison and Yao, 2016). But property boundaries may affect road construction and influence costs. The FIF assumes close forests can share harvest road construction costs. However, in reality forests may be different ages and this might reduce the incentive to share costs.

The feature that differentiates FIF from the other economic frameworks is the financial viability component that uses long-term forest productivity surfaces combined with spatially explicit forest management costs. The original Forest Investment Finder only included timber and carbon sequestration values. Then an avoided erosion layer was added, and it was termed FIF plus (Barry et al., 2014). Following this, further spatial layers were developed such as avoided nutrients and it is now referred to as the FIF. Since 2014, more than 21 case studies have been undertaken across New Zealand using the FIF (Yao et al., 2021) although many results are confidential (see Yao et al., 2019 for a list of some specific applications of the FIF). The FIF uses have included: (1) assessing the afforestation potential of marginal land; (2) support the modelling of policy scenarios focusing on regional economic development; (3) assess both market and non-market values of key landscape areas; and (4) provide a clearer description of the complementarity of planted forests with other productive land-uses. However, it does not appear to have yet been used to assess the economic (or environmental) impact of various forestry management/environmental mitigation practices.

The inputs and outputs in the FIF are described in Figure 4.

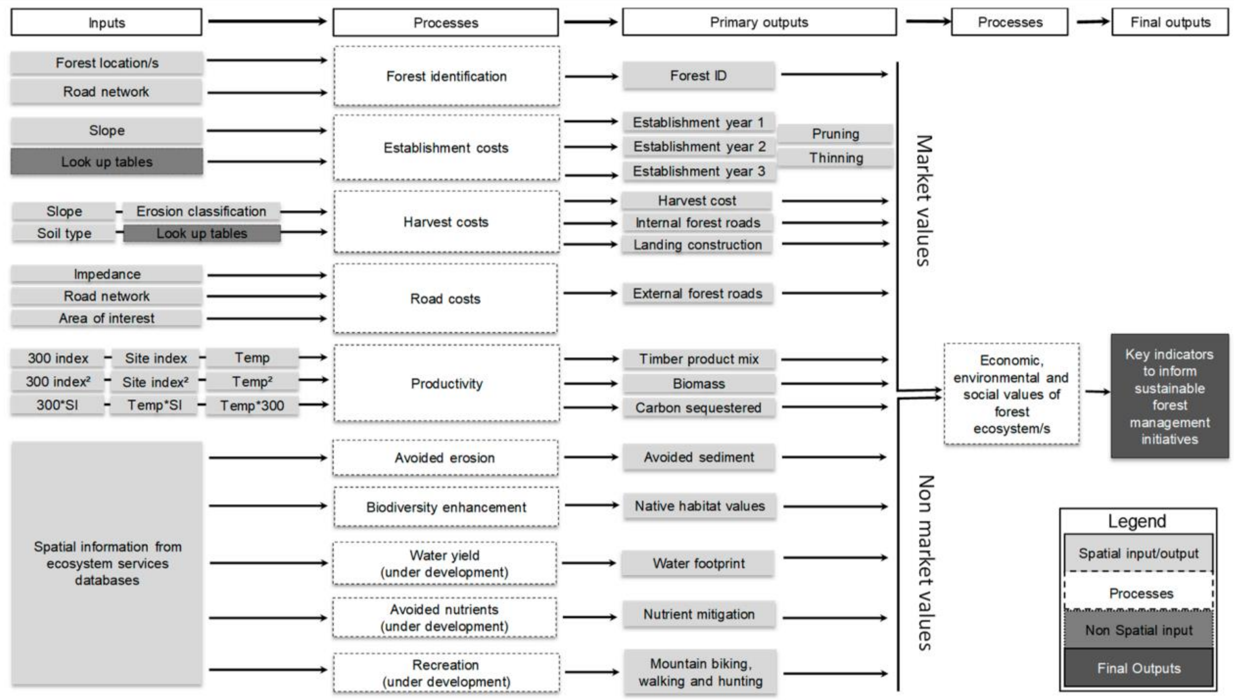

Figure 4: Diagram of FIF data inputs, processes and outputs (Yao et al., 2019).

The key highlights of the FIF as applicable to this report are noted below, these are derived from Yao et al. (2021), Harrison et al. (2012) and Hock et al. (2016):

- The returns from forestry are estimated by providing predictive surfaces for log volume, tons of biomass, and carbon sequestration. This is done by assigning candidate areas with information from underlying spatial surfaces that describe the potential productivity, as well as factors that contribute to forest management costs such soil type, slope of the terrain, and distance to markets.
- The FIF productivity metrics are based on the 300 Index and the Site Index (Palmer et al., 2010, 2009). The 300 Index is an index of volume mean annual increment, and Site Index measures height at a reference age (see Palmer et al. (2009) for details).
- Timber viability is assessed by assuming a *Pinus radiata* structural (framing) or unpruned regime (thinned to 600 stems / ha from initial planting of 900 stems / ha) with a rotation length of 28 years.
- To calculate the log revenue, different log grades are multiplied based on the productivity layer by average log prices reported in AgriHQ.
- To calculate the cost of producing logs, the FIF uses spatial cost functions for plantation establishment, silviculture management regimes, landing development, road construction and maintenance, log harvesting operations, and transport of logs to the point of sale (e.g., shipping port or timber mill). Costs are calculated based on average costs for New Zealand reported in databases (e.g., AgriHQ).
- Impedance factors, such as slope and soil type, affect the cost of forest operations meaning that the FIF can, for example, account for the impacts of steeper slopes on harvesting cost as well as road construction and maintenance. Hock et al. (2016) specifically included impedance costs of road construction based on erosion, slope and rainfall values as well as for roads that traverse streams and rivers.

- A slope adjustment factor, based on how difficult it is to travel across a site, is used to modify establishment and silviculture costs within the framework. The slope adjustment factor is 1 for flat land (0-5 degree slope), 1.08 for rolling land (5-15 degree slope), 1.25 for steep land (15-25 degree slope) and 1.72 for very steep land (more than 25 degree slope).
- Establishment costs include site preparation, cost of seedlings, cost of planting labour, weed control, and other inputs incurred in the first three years of the rotation. Cost of seedlings varied across studies, \$250/1000 *Pinus radiata* seedlings and \$390/1000 Douglas fir seedlings in Harrison et al. (2012), and \$400/1000 *Pinus radiata* seedlings in Hock et al. (2016). See Table 16 for planting costs.
- Silviculture costs include the cost of thinning and pruning which are incurred between years 5 and 11. Costs for silviculture practices are estimated using labour costs, which include the costs of chainsaws, fuel, protective clothing transport and overheads, and the time taken to perform a certain task such as thinning and pruning. Task times are derived from relevant silvicultural time standards. See Table 17 for thinning costs.
- Harvesting costs include felling, logging system (skidder, hauler, cable), cutting of logs, sorting according to log grades, quality control and loading onto a logging truck. They are summarised in Table 18.
- Costs for landings and forest roads are calculated using landing and road density estimates. The density at which landings and roads occur within a forest are assigned to various slope classes.
- The calculation of transport costs from the forest location to the destination (port, sawmill, processing plant) are undertaken on a distance basis. The cost to transport a ton of timber multiplied by distance in kilometre based on a rate of \$ 0.22 per tonne per kilometre. It is assumed in Harrison et al. (2012) that all S1, S2 and pulp is sent to the mill/processing plant and all S3 logs are sent to the port. The reason being that approximately 45% of the total timber is exported, and s3 logs are approximately 45% of the total log grade mix.
- FIF can also be used to determine whether a new road is required to reach each forest location and estimate its cost of construction. FIF can be used to identify the least cost path to construct a road from the current road network to the forest. The impedance surface assigns cost associated with building a road across slope, erosion, and rainfall classes.
- The FIF tool also includes costs incurred when crossing a water way (e.g., river, stream). When modelling road construction, small streams incur a relatively low cost, compared to rivers and wetland areas that are given prohibitively large costs to ensure avoidance during cost path model calculations.
- Data have been incorporated into FIF so that it can be used to account for opportunity costs by including the value of the land of interest or, in other applications, the potential land rental values.
- The current version of the FIF in this validation exercise uses a national average cost and does not account for variation that may exist across New Zealand regions.

Table 16: Planting costs for forestry regimes using the FIF, adapted from Hock et al. (2016) and Harrison et al. (2012).

Slope	Cost of planting *Pinus radiata* (structural regime) from 900 stem/ha (Hock et al., 2016) $/ha			Cost of planting *Pinus radiata* (general unpruned regime) at 1200 stem/ha (Harrison et al., 2012) $/ha			Cost of planting Douglas fir (general unpruned regime) at 1600 stem/ha (Harrison et al., 2012) $/ha		
Year	1	2	3	1	2	3	1	2	3
0-5	1,081.42	224.90	224.90	1,260	250	58	1,585	250	58
5-15	1,114.95	235.77	235.77	1,296	305	62	1,620	305	62
15-25	1,193.20	261.13	261.13	1,380	331	66	1,704	331	66
>25	1,403.35	329.23	329.23	1,604	399	79	1,927	399	79

Table 17: Thinning costs for forestry regimes using the FIF, adapted from Hock et al. (2016) and Harrison et al. (2012).

Slope	Cost of thinning *Pinus radiata* (structural regime) from 900 to 700 stem/ha (yr 7) (Hock et al., 2016) $/ha	Cost of thinning *Pinus radiata* (general unpruned regime) from 1200 to 500 stem/ha (yr 9) (Harrison et al., 2012) $/ha	Cost of thinning Douglas fir (general unpruned regime) from 1600 to 600 stem/ha (yr 15) (Harrison et al., 2012) $/ha
0-5	169.56	658	960
5-15	174.72	672	980
15-25	186.78	720	1,030
>25	219.16	810	1,160

Table 18: Logging costs ($/tonne) by terrain system and location in FIF, adapted from Harrison et al. (2012)

Slope	Island	Extraction type	Agrifax value ($)
0-5	North Island	Flat Ground-based	21
5-15	North Island	Tracked Ground-based	26
15-25	North Island	Steep Tracked	30
>25	North Island	Hauler	39
0-5	South Island	Flat Ground-based	26
5-15	South Island	Tracked Ground-based	26
15-25	South Island	Steep Tracked	31
>25	South Island	Hauler	33

Alternatively, for very small forests (e.g., woodlots on farms that are less than 100 ha), accurate timber viability predictions would require a stand level timber growth and yield software tool (e.g., Forestry Forecaster; West et al. (2013)). However, to be able to use Forecaster, the analyst requires a forestry background and ideally on-site tree measurement data for accurate estimates of timber yields from forest plots (West et al., 2013).

3.3 Water quality impact

Management actions simulated for rural HRUs in the FWMT result in changes to total N, total P, total suspended sediment and *E coli* discharged to freshwater (as well as changes to simulated hydrology).

Orders of magnitude greater concentrations of *E. coli* are generally found in streams running through pasture, due to animal effluent discharge, than streams in forested catchments (Donnison et al., 2004; Parkyn et al., 2006; Young et al., 2005). Therefore, this section presents information on the base contaminant loss from forestry of N, P, and sediment. This section presents both quantified information where possible, as well as qualitative information, for parts of the forestry cycle. The section excludes discussion of hydrological changes, which are pronounced during the forestry cycle.

Numerous biophysical and management factors affect the environmental footprint of plantation forestry, such as underlying geology, hydrology, previous land use, stand age, and forest management and harvesting practices. Underlying natural processes and legacies from historical land-use practices can sometimes have a greater influence on some aspects of water quality in planted forests than the current land use itself (Davis, 2014; Parkyn et al., 2006; Quinn et al., 1997).

Mature planted forests often have water quality attributes similar to those in undisturbed indigenous forests, but water quality can vary considerably throughout the forest cycle depending on the management activities and the level of disturbance they generate (Harding and Winterbourn, 1995; Hartman, 2004; Neary et al., 2011; Pike et al., 2010). In particular, there is a 'window of vulnerability' which refers to the time between trees being removed when roots slowly decay and soil reinforcement is reduced and is not fully compensated for by the replanted trees for several years following planting. In addition, the removal of trees allows soil moisture conditions to be wetter for longer (loss of interception capacity of canopy and evapotranspiration), which contributes to this high-risk period. Timber harvesting changes the two mechanisms that provide stability (hydrological and mechanical) and in turn changes the threshold conditions for erosion failure across the landscape. The window of vulnerability for New Zealand plantations is estimated to be in the period 1-6 years (Figure 5) after harvesting but is species and density dependent (Phillips, Marden and Basher, 2012).

This window of vulnerability is particularly relevant for erosion (and associated contaminant loss). In the window of vulnerability, soil, slope and geomorphic conditions are such that even without a "big" trigger such as a severe rainstorm, mass movements are highly likely to result. If a severe storm was to occur, then the degree of damage/land sliding would be elevated because a wider range of slope threshold conditions will be met (Phillips et al., 2012). Around 20% of the national planted forest estate is in steep (>20°) hill country which is highly erodible and susceptible to extreme weather events (Dunningham et al., 2012).

Nutrient losses are also linked to the window of vulnerability. Knowles, Hansen and Laroze (2004) do not provide quantification of nutrient losses but summarises the nutrient cycle across the forestry rotation. In the establishment phase (0-8 years), nutrient uptake by the young trees is high since they build up biomass and expand their crowns rapidly. Between crown closure and harvest (about 8 –28 years for *Pinus radiata*), nutrient uptake is reduced (Quinn and Ritter, 2003). The most critical phase with respect to water quality will be the period immediately following harvesting when tree cover is removed and subsequently leaching of nutrients and erosion can increase again.

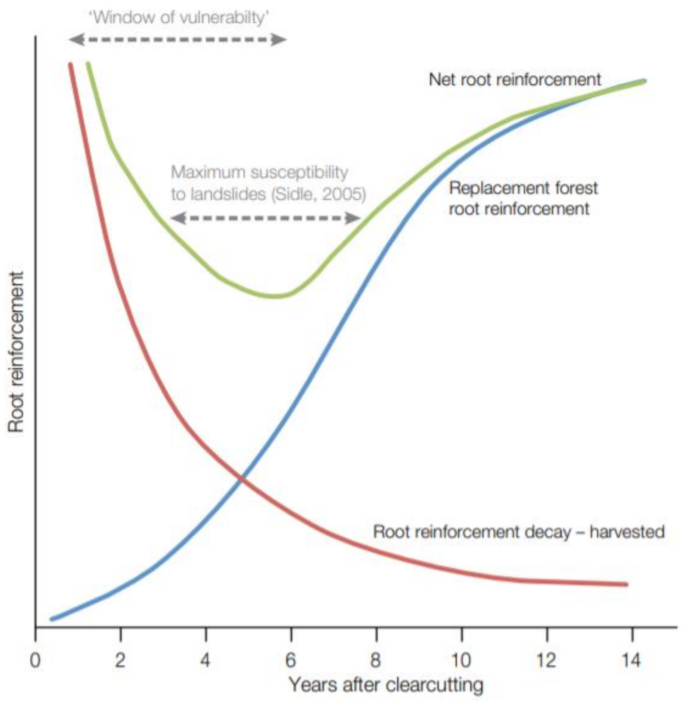

Figure 5: An example of typical changes in forest vegetation root strength or root reinforcement after timber harvesting showing the 'window of vulnerability' (adapted from Phillips, Marden and Basher, 2015).

3.3.1 Nitrogen

The extent of nutrient losses in forestry depends on biophysical factors, harvesting techniques, fertiliser applications, silvicultural practices (e.g., weed control, pruning, thinning) (Payn and Clinton, 2005), the history of the land, management of the forest, and time since planting (Monaghan, Semadeni-Davies, Muirhead, Elliott and Shankar, 2010) and will vary across the forestry cycle for plantation forestry. The most critical phase in the forestry cycle is the window of vulnerability. Quinn and Ritter (2003) found that nutrient concentrations in streams quickly decline back to pre-harvest levels again following replanting and even a decrease in N leaching after clear-felling has been reported (Parfitt et al., 2002). Forests are conservative in cycling N and little is lost through leaching (Payn and Clinton, 2005). Nutrients are returned to the soil from the canopy as litterfall or deposited as thinning or harvesting slash. These materials decompose slowly and N is released mainly as ammonium N (Parfitt et al., 2001; Girisha et al., 2003; Will et al., 1983). Some leaching loss of N from forests may occur, especially at harvest, when N uptake is disrupted and decomposition accelerated.

Ledgard (2014) estimated N loss figures for land uses in the Southland region based on a literature review. They found forestry loss was 2 kg N/ha/yr with a range of 0.5 kg N/ha/yr to 5 kg N/ha/yr (Ledgard, 2014). It is assumed that this is averaged over the course of the rotation. Davis (2014) noted that while N losses from forests are generally low and between 0.25 and 2.5 kg N/ha/yr, application of N fertiliser and other management practices have the potential to increase leaching losses of nitrate and other forms of N from forests. This aligns with Ledgard (2014). Similarly, Matheson et al. (2018) estimated N losses to water at 2.5 kg N/ha/yr from exotic plantation forest as an average annual loss over the course of rotation using Overseer. The Overseer model estimates N losses to water at 3 kg N/ha/year from native forest. Hansen, Knowles and Halliday (2004) estimated the N loss from plantation forestry as a base loss of 2 kg N/yr and SCION (2019) estimated N loss for both indigenous forests and planted forests without fertiliser addition at 0.25 to 2.5 kg/ha/yr, and up to 5 kg/ha/yr in areas with volcanic soils. It is assumed that this is averaged over the course of the rotation. Yao et al.

(2021) used the N leaching rate under forestry of 3 kg N/ha/yr based on the work by Yao and Velarde (2014). PCE (2004) estimated annual loadings of N to surface waters from non-point sources of 100,000 t/yr (agriculture), natural forests 15,000 t/yr and plantations 7,000 t/yr or, 8, 4, 2 kg N/ha/yr, respectively. There is some evidence of higher losses in other regions. Monaghan et al. (2010) suggested that *Pinus radiata* planted on improved pasture in the Lake Taupo catchment may lose between 8 and 12 kg N/ha/yr, likely due to the soil type.

Dyck, Gosz and Hodgkiss (1983) also compared N loss under a range of different forest systems based on suction cup measurements on pumice soils. These are summarised in Table 19.

Table 19: Summary of average N concentrations in suction cup solutions from undisturbed forest and gorse on pumice soils, adapted from Dyck et al. (1983).

Forest	Location	Stand age	Stocking rate (stems/ha)	Soil	N (kg/ha/yr)
Pinus radiata	Kaingaroa	20	250	Silty sand	0.03
Douglas fir	Kaingaroa	56	200	Sand	1.2
Eucalyptus saligna	Rotoehu	18	100	Coarse sand	0.36
Gorse	Rangitaiki	20	Dense	Sand	22.9

Table 20 provides a summary of data on N removals (kg/ha) for stem harvests for a range of sites, adapted from Payn and Clinton (2004). These are not annual estimates.

Table 20: Summary of data on N removals for a variety of stem harvests for a range of sites, adapted from Payn and Clinton (2005). The annual value is simply averaged across the stand age.

Source/ location	Soil type	Age	Species	kg N/ha	Kg N/ha/yr
Webber & Madgwick 1983		29	*Pinus radiata*	217	7
Berwick Forest	Mottled Fragic Pallic Soil	31	*Pinus radiata*	337	11
Canterbury Plains Forest	Pallic Orthic Brown Soil	32	*Pinus radiata*	187	6
Golden Downs Forest	Acidic Orthic Brown Soil	45	*Pinus nigra*	280	6
Kinleith Forest	Immature Orthic Pumice Soil	26	*Pinus radiata*	136	5
Tarawera Forest	Buried-Pumice Tephric Recent	27	*Pinus radiata*	187	7
Woodhill Forest	Typic Sandy Recent Soil	42	*Pinus radiata*	128	3

A number of studies looked an N loss at different times in the forestry. Quinn and Ritter (2003) noted that the loss from plantation forests in the four years following land-use conversion was twice the base leaching (i.e., 4 kg N/ha/yr). Similarly, it was assumed that the loss doubled in the year following clear-felling (Quinn and Ritter, 2003). Yao et al. (2021) suggested that while the average N leaching rate of 3 kg N/ha/yr applies for undisturbed NZ forests and this rate can increase to 28 kg N/ha/yr at harvest of *Pinus radiata* forest in a particular NZ region based on Menneer, Ledgard and Gillingham (2004).

Dyck (1982) reported leaching losses of up to 15 kg N/ha/yr for a two year period following harvest at Kaingaroa forest. This shows that when plant uptake is low or removed, mobile nutrients can become available for leaching. Dyck et al. (1983) used suction cups at soil depths of 1 m and found that one year after harvesting operations on a volcanic ash soil in Kaingaroa Forest, concentrations of nitrate in leachate increased nine-fold from 0.07 to 0.60 mg/L. Assuming annual rainfall of 1,500 mm and drainage of 450 mm, these values would translate to an increase in leaching from 0.3 to 2.7 kg N/ha/yr. The elevated levels persisted for three years after logging.

In terms of N loss across a full rotation of plantation forestry, Quinn (2003; 2005) provided some data for a high nitrogen (fertilised former pasture) site; Puruki, a *Pinus radiata* forest in the Purukohukohu experimental basin between Lakes Taupo and Rotorua. Quinn (2003; 2005) found that N concentrations in stream water showed a gradual decline over the first five years after conversion from pasture to pine with the average N concentration being almost an order of magnitude lower in the young forest phase (age 6–13 years), but levels then increased, indicating that the forest, at this high-N site, started to leak nitrate as it matured. When harvested, N yields increased to be similar to those in an adjoining catchment in pasture, but then declined rapidly to be similar to yields in the young forest phase. Parfitt et al. (2002) had found a similar pattern in a sub-catchment of the Puruki catchment. This is shown in Figure 6. While harvesting can lead to a short-term increase in N leaching, it is not clear how common this is given a lack of studies. A large (345 ha) catchment (Pakuratahi) study located in the coastal hill country of Hawke's Bay showed no increase in N or P levels in stream water for a period of six years following harvest, despite the harvest being concentrated over a two-year period (Fahey and Stansfield 2006). Increases in nutrient leaching following harvest might, therefore, be restricted to areas of high fertility or where revegetation has been suppressed.

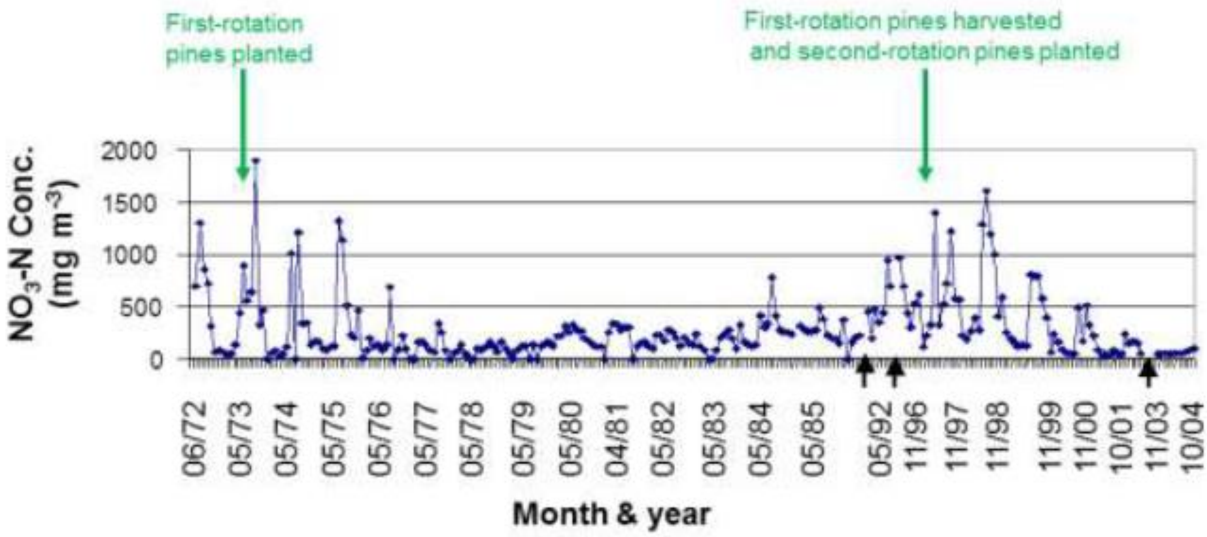

Figure 6: Long-term variation in N (nitrate) concentration in stream flow in Puruki catchment from pasture prior to pine planting (1973) through to logging and replanting (1997) and regrowth of the second pine crop. Short arrows indicate when data was not collected (from Quinn, 2005)

Davis (2015) utilised the same data from the Puruki catchment and translated this into N loss data across the forestry cycle. This is shown in Figure 7. The N leaching loss from the pasture prior to planting was 5.7 kg N/ha/yr and increased to 11.1 kg N/ha/yr in the year following planting before declining to pre-planting levels in year two. The leaching loss then declined to 1-2 kg N/ha/yr between years three and five and to less than 1 kg N/ha/yr between years 6 and 14. Records were not collected between years 15 and 23. Nitrogen leaching losses were higher from the mature pine forest, just before logging, than in the young forest phase (years 3-14), suggesting that N retention by the pine forest is greatest when the pine crop has established and is growing vigorously, but the forest becomes more prone to nitrate loss by leaching as it matures (Quinn and Ritter, 2003). Nitrogen losses in the year of logging and the first year after logging increased to 3.9 and 4.5 kg N/ha/yr respectively but decreased rapidly toward levels seen during the young forest phase which is likely due to weed growth (Parfitt et al., 2002).

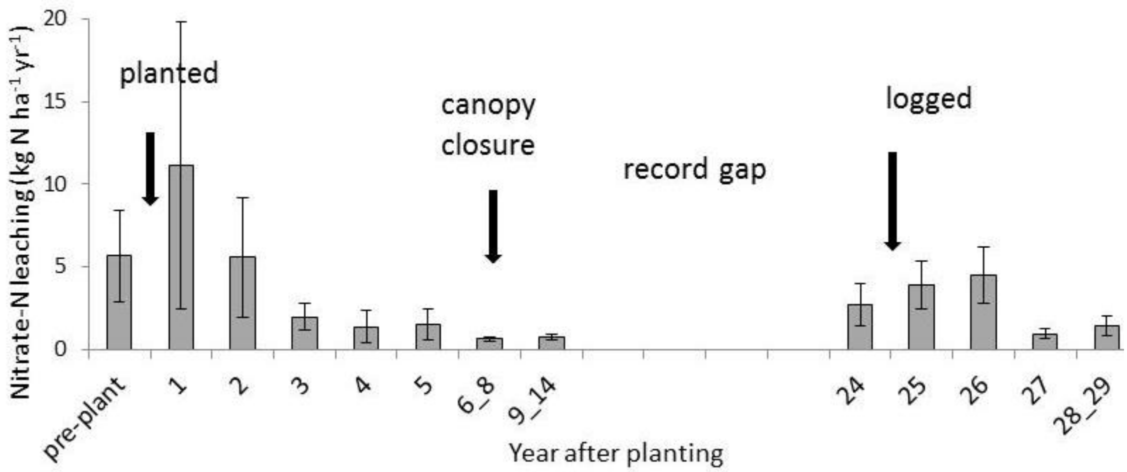

Figure 7: Long term variation in N leaching losses from Puruki catchment of pasture before *Pinus radiata* planting, through tree planting, growth and logging to regrowth of the second rotation crop. Bars show standard errors, sourced from Davis (2014).

Davis (2014) notes that there could be a difference in N loss from plantation forests based on the prior land use (i.e., pastoral or previous forestry land uses). Average N leaching estimated from lower root zone soil water samples from planted forests on previous non-agricultural land was 3.25 kg N/ha/yr. Comparatively for an ex-farm site at Massey, Manawatu where total mineral N leaching losses declined rapidly from 18 kg N/ha/yr in the year of conversion of pasture to young forest, to less than 1 kg N/ha/yr in the second year. By year nine, after canopy closure and N demand by the tree crop had declined, leaching had increased to 4.5 kg N/ha/yr (Parfitt and Ross 2011). A similar transition occurred at age ten in the second rotation of a site at Tikitere, Bay of Plenty, where nitrate-N leaching increased from 0.2 kg N/ha/yr in the first 11 months of the sampling period to 28.7 kg N/ha/yr in the following 18 months (Davis et al., 2012).

3.3.2 Phosphorus

Forestry blocks can be sources of P loss to water. Phosphorus losses from pine catchments are usually higher than those from native forest catchment but lower than losses from pasture (Monaghan, et al., 2010). There are not many studies on P loss available to inform P loss estimates in forestry.

Ledgard (2014) estimated P loss figures for different land uses in the Southland region based on a literature review. They found P loss from forestry was 0.2 kg P/ha/yr with insufficient information to present a range. This estimate was based on Monaghan et al. (2007). It is assumed that this is averaged over the course of the rotation. Similar to this, Matheson et al. (2018) used Overseer to estimated P losses to water at 0.1 kg P/ha/year from exotic plantation forest as an average annual loss over the course of rotation. The Overseer model estimates P losses to water at 0.1 kg P/ha/year from native forest.

Table 21 provides a summary of data on P removals (kg/ha) for stem harvests for a range of sites, adapted from Payn and Clinton (2005). These are not annual estimates.

Table 21: Summary of data on P removals (kg/ha) for stem harvests for a range of sites, adapted from Payn and Clinton (2005). The annual value is simply averaged across the stand age.

Source/location	Soil type	Stand age	Species	P	kg P/ha/yr
Webber & Madgwick 1983		29	*Pinus radiata*	31	1.1
Berwick Forest	Mottled Fragic Pallic Soil	31	*Pinus radiata*	38	1.2
Canterbury Plains Forest	Pallic Orthic Brown Soil	32	*Pinus radiata*	28	0.9
Golden Downs Forest	Acidic Orthic Brown Soil	45	*Pinus nigra*	67	1.5
Kinleith Forest	Immature Orthic Pumice Soil	26	*Pinus radiata*	22	0.8
Tarawera Forest	Buried-Pumice Tephric Recent	27	*Pinus radiata*	36	1.3
Woodhill Forest	Typic Sandy Recent Soil	42	*Pinus radiata*	38	0.9

3.3.3 Sediment

Sediment loss from forestry can vary across the rotation and be significant at times, especially harvesting and during the window of vulnerability. Sediment loss is focused on certain times within the rotation, particularly construction of infrastructure (roads) and harvest. Sediment yields elevated by harvesting activities typically return to pre-harvest levels, or decline markedly, within two to six years of harvest (SCION, 2017a). A lot of the studies to date are qualitative and relate forestry as a land use to pasture. There is not robust information quantifying the relative sediment yields from different activities within a forestry system, across different biophysical characteristic (e.g., soil types).

Sediment production is a function of the periodic removal of forest cover and the gradual decay of root systems, which predispose soils to greater erosion risk. The risk of soil loss by erosion is most pronounced in the five-to-eight-year window of vulnerability between the decay of harvested tree root systems and the establishment of the next tree crop. Studies of hill country farms on the East Coast (McElwee, 1998) and in Hawke's Bay (Halliday and Knowles, 2003) have shown that tree roots have a vital role in holding erosion prone soils onto steep slopes.

A key challenge with managing sediment is managing the incidence of weather events when they combine with harvest. Managing post-harvest negative impact events is not an easy task, the challenge is to manage for their occurrence, adopt any measures that could reduce their number, and reduce the impacts they have on downstream environments (Phillips, Basher and Marden, 2016). This influence of weather events on sediment losses can be challenging to study and can make comparison between studies difficult.

Sediment can also contribute to debris flows, which are a 'very rapid to extremely rapid surging flow of saturated debris in a steep channel' (SCION, 2017b). Importantly they are distinguished by very high sediment concentrations by weight, about 80% (SCION, 2017b). They may or may not contain woody material from forests (i.e., they can be sediment-only). The primary cause of debris flows is the generation of sediment by mass movement into a channel. These lead to one-off sediment loss events and can be challenging to model. Landsliding and debris flows, and therefore sediment loss, are typically triggered when a rainfall threshold is exceeded. This makes modelling sediment loss challenging as it requires predictions of rainfall thresholds, timing of rainfall events etc.

Harvesting in steep hill country often increases sediment yields with most sediment reaching waterways via run-off from roading activities, landslides, and channel-bed scouring with minor contributions from slope erosion (Fahey and Marden, 2006; Fransen et al., 2001; Marden et al., 2006).

Sediment yields elevated by harvesting activities typically returned to pre-harvest levels, or declined markedly, within two to six years of harvest (Basher et al., 2011; Fahey and Marden, 2006; Phillips et al., 2005). Within a logged area, sediment can be generated both as a consequence of management practices during harvesting and natural processes after harvesting (e.g., erosion through rain and storm-initiated landslides). Sediment can also be generated during the pre-logging phase of road and landing construction.

Mature plantation forests typically have lower erosion (surface erosion) and land sliding than other times in the plantation forestry cycle (Phillips et al. 2012). Forestry can contribute significant amounts of sediment at particular stages of the production cycle. A Nelson study found that when road construction was at its peak, the sediment mobilisation rate was about 3,200 kg/ha/yr; this subsequently decreased to around 10% of this figure (Fahey and Coker 1989). Not all of this sediment was predicted to reach a stream. Similarly, Marden et al. (2007) studied slopewash in pumice terrain and found sites with deep disturbance generated 3,800 kg/ha, but by 21 months after harvest when groundcover occupied 80% of plot area, sediment generation had declined to almost zero.

Basher (2013) noted that the impact of sheet erosion (and therefore, associated sediment loss) occurs more on areas of bare ground. This includes forestry cutovers (Marden and Rowan 1997; Phillips et al. 2005; Marden et al. 2006, 2007), unsealed roads and tracks (Fahey and Coker 1989, 1992) and earthworks associated with farming, forestry or other land uses (Hicks 1994).

Fahey et al. (2003) separated out the sediment yields for the road construction phase noting that this was a time often associated with high sediment yields. While the results are presented in Table 22 the study also describes the type of roading used. However, there is limited option to compare this to other forestry roading systems or if a different roading configuration had been used. Roading in this study included extending existing roads, adding 3.5 km of new road, 25 new hauler pads and the upgrading of 20 hauler pads. Coincidental with the post-harvesting period, a series of moderate-sized storms were identified as the likely sources of increased stream sediment yield in the Pakuratahi (Fahey et al. (2003) although ground disturbance during hauler-logging undoubtedly contributed to the increase in suspended sediment yield.

The suspended sediment yield following logging of a 36 ha plantation forest catchment in Coromandel was determined over a 30 month period to March 2003 in Phillips et al. (2005). Total suspended yield for the catchment for the whole period was 73.2 t, and on an annual basis this ranged between 59 and 116 t km^{-2} (Phillips, et al., 2005). Two storms in April 2001 and February 2002 contributed 37% of the total sediment yield (Phillips, et al., 2005). This indicates the importance of storm events in sediment generation.

Table 22: Suspended sediment yields for storms monitored concurrently at Pakuratahi (forest) and Tamingimingi (pasture) catchments, and the ratio of sediment yields (July 1995-July 2000), adapted from Fahey et al. (2003)

Period	No. of storms	Pakuratahi Yield (t/km^2)	Tamingimingi Yield (t/km^2)	Ratio (Pak:Tam)
Pre-harvesting (Jan. 1995 to June 1997)	9	7.6	21.2	1:2.8
Road construction phase (July to Dec. 1997)	3	4.9	22	1:4.5
Logging phase (Jan. 1998 to Dec. 1999)	10	84.4	43.8	1:0.52
Post-harvesting (Jan. 2000 to Dec 2001)	5	176.84	129.05	1:0.73

While differences in sediment concentrations were evident among land uses (as in Fahey et al., 2003), this is influenced by underlying geology, slope stability and storm patterns during the study periods (Baillie & Neary, 2015). Baillie and Neary (2015) summarised a range of sediment related indicators across plantation forest (mature), indigenous forest and pasture, these are presented in Table 23. It is unclear if 'mature' plantation forest includes the expected sediment yields across the whole forestry cycle (i.e., includes forest harvest).

Table 23: Comparison of sediment attributes among streams in different land uses, adapted from Baillie and Neary (2015)

Land use	Turbidity (NTU) range	Suspended sediment (g/m^3)	Sediment yield ($t/km^2/yr$)
Mature planted forest	0.8 – 100	0.5 – 100	1.7 – 65
Indigenous forest	0.3 - 3.5	0.5 – 2.5	5.1 – 160
Pasture	1 – 580	2 – 700	4.3 – 106

Relative to pasture, forestry provides erosion benefits including, a reduction in shallow landsliding, reduced rates of earthflow movement, reduced gully erosion and the retention of soil (Marden & Rowan 1993; Phillips & Marden 2005; Marden 2005). However, it is important to remember that this is considered across the whole forestry rotation and there are high risk periods within the forestry cycle including harvesting and the window of vulnerability. Long-term findings across the forest cycle are not necessarily indicative of differences in event-based loads, particularly during the window of vulnerability (post-harvesting period).

There is not extensive robust information quantifying the relative sediment yields from different forestry activities across different biophysical characteristic (e.g., soil types, geology and slope). Fahey, Marden and Phillips (2003) is one of the limited studies there is. Fahey et al (2003) looked at some of the key sediment generating phases for a forested catchment in Hawkes Bay, this is summarised in Table 22. Fahey et al. (2003) calculated suspended sediment yields for over 50 events from 1995 in two catchments in erodible hill country in coastal Hawke's Bay; one in pasture (Tamingimingi, 7.95 km^2) and the other in initially mature *Pinus radiata* plantation forest (the Pakuratahi 3.45 km^2). The pine forest was harvested by skyline hauler (85%) and skidder (15%). Post harvesting treatment included oversowing with grass and legumes and ripping of hauler pads before replanting. Pre-harvesting total suspended sediment yield for the pasture catchment was three times higher than for the catchment in mature pines. In the logging phase of the harvesting period the situation was reversed with the total yield for the harvested catchment twice that of the one in pasture (Fahey et al., 2003). Over the seven years of data the catchment originally planted in pines and subsequently harvested has yielded only 20% more suspended sediment than the one in pasture.

Ritchie (2011) also highlights sediment yields in paired studies as shown in Table 24. Based on this, it appears that pasture slopes generate 2-5 times more sediment than comparable forestry slopes, except during forestry harvest periods (Ritchie, 2011).

Table 24: Comparison of sediment yields in different land uses in paired studies, adapted from Ritchie (2011)

Study	Scenario	Sediment yields	Approximate ratio
Quinn & Stroud (2002): Hill country (Whatawhata)	Pasture	998 kg/ha/yr	3.1:1
	Bush	320 kg/ha/yr	
Dons (1987): Pumice soil (estimated)	Pasture	220 kg/ha/yr	5.5:1
	Mature pines	40 kg/ha/yr	
Fahey & Marden (2000): Hawkes Bay (estimate of total in 29 months)	Pasture	1,044 kg/ha	3.2:1
	Mature pines	327 kg/ha	
Fahey et al. (2003): Hawkes Bay (storm yields measured concurrently) Pre-harvest (9 storms over 29 months)	Pasture	212 kg/ha	2.8:1
	Mature pines	76 kg/ha	
Fahey et al. (2003): Hawkes Bay (storm yields measured concurrently) Logging period (10 storms over 2 years)	Pasture	439 kg/ha	1:1.9
	Forestry logging	844 kg/ha	
Fahey et al. (2003): Hawkes Bay (storm yields measured concurrently) Forestry post-harvest period (5 storms over 2 years)	Pasture	1,290 kg/ha/yr	1:1.4
	Forestry post-harvest	1,768 kg/ha/yr	
Fahey & Corker (1989): Sediment mobilised from forestry tracks (Nelson)	Not paired study	3,200 kg/ha/yr	NA
Philips et al. (2005): Sediment mobilised by forestry harvest (Whangapoua)	Not paired study	590-1,160 kg/ha/yr	NA

Mature, closed-canopy, indigenous or exotic forest (and scrub) typically reduces landsliding by 90% relative to unforested land, and has been used to control severe gully erosion and reduce rates of earthflow movement (by 2–3 orders of magnitude) (Basher et al., 2019). Trees younger than about 8 years, before canopy closure, are far less effective in reducing erosion. Basher et al. (2019) noted that this effectiveness of afforestation relative to pasture is 80% for plantation forestry, rather than permanent forest. They do note that this takes approximately 10 years to reach 'maturity' of effectiveness.

While the effects of forest harvesting on increasing sediment yield, and the consequences of poor road and landing construction and maintenance have characterised in some early studies (e.g., Pearce & Hodgkiss 1987; Fahey & Coker 1989, 1992) little is known of the performance of modern engineering standards for water control, road and landing construction (Basher et al., 2019).

About one-third of the New Zealand plantation forest estate is located on erodible steeplands, with many of the forests having originally been planted as erosion control forests. It is acknowledged that it will be impossible to completely avoid slope failures and debris flows following harvesting (Phillips et al., 2012), and the focus needs to be on risk assessment to manage the incidence and consequences of these events (Basher et al., 2015).

3.4 Summary

This section reviewed the literature on the base state of forestry, including both the economic and environment performance. This is currently intended for use in supporting the calibration of the FWMT, although consideration is given to if this would be suitable for use in future scenario modelling.

It is recommended that the model focuses on *Pinus radiata* at this stage, harvested at 27 years. At this stage, it is likely that base profit and environmental footprint data can be isolated for farm plantation forest (small scale plantation forest within a farm boundary), plantation forest and mixed forest (native) categories as defined in section 1.3. It is acknowledged that these categories are slightly different to the base HRU types, however, are based on what is most appropriate given the base and mitigation data available.

3.4.1 Baseline profit estimate

It is recommended that at this stage the FWMT use an EAA for profit. This is basically the NPV on an annualised basis. Some data will need to be adjusted, while data that cannot be adjusted cannot be used. However, the EAA enables data to be included into the current FWMT format.

The profitability of forestry depends on what is included in the returns, and the primary income streams at the moment are timber and carbon. Throughout the research there is a wide range of profitability, different metrics are used, and it is difficult to separate out the NPVs in research into component parts. It would be desirable to use a model such as FIF to generate profitability metrics for the different typologies that are specific to the Auckland region and are underpinned by discussions with foresters in Auckland. However, in lieu of this, farm plantation forestry could be assumed to have an annual profit of $185 /ha, plantation forestry an annual profit of $430/ha and permanent forest of $500/ha. The higher profitability of permanent forest reflects the increasing carbon price.

The farm plantation forestry profit estimate is derived from the average of the relevant Perrin Ag One Billion Trees project case studies and Moore et al. (2015) (Table 9 and Table 10). This includes mostly *Pinus radiata* and includes carbon, a 6% discount rate was used and with the information available an equivalent annual annuity was able to be calculated. While the NZFFA data source provides detail on nominal net annual returns that can be isolated to the Northland area, the nominal net annual return cannot be adjusted to an EAA for use in LCC model which feeds into the FWMT. To adjust the nominal net annual returns to an equivalent annual annuity there would need to be an estimate of when costs and benefits were incurred/received to enable discounting.

The annualised profit estimate for the plantation forestry is based on an average of all the literature that is applicable, is in an equivalent annualised annuity or able to be converted to one (i.e., that has an NPV, discount rate and specified time-period). The studies ranged from $35/ha/yr to approximately $600/ha/yr. The studies ranged in their inclusion of carbon or not, carbon prices, differential carbon sequestration rules, location, time-period and forest management. All were predominantly *Pinus radiata* and approximately 27-year rotations. Most use the averaging approach to carbon income, or equivalent, where carbon income is earned in the first 16 years with no obligation to repay any credits so long as the forest is replanted after harvest. Given the proximity of the Auckland region to port facilities it is likely that forestry in Auckland is one of the more profitable regions (as seen in the NZFFA (2019) data). This means the estimate of the annualised annuity derived from studies across the country could be lower than for Auckland forests; however, this cannot be validated using a literature review approach.

The permanent forest profitability estimate is based on permanent forest and therefore is based on carbon income only. This is based on the analysis in Harrison and Bruce (2019), this study was based in Wairoa, over a 60-year period with a 5% discount rate and a carbon price of $25/t. A range of carbon values have been used in the studies that inform these estimates. The carbon price will likely change over the 50 years involved in the LCC for the FWMT. This is best explored in a sensitivity analysis which could be completed in specific modelling exercises.

Environmental premiums are considered out of the scope of this project. While there are some suggestions that when products from the primary sector, including forestry, meet particular environmental thresholds they will receive additional price premiums, there is little evidence of this happening in practice currently (although this may change in the future, e.g., through the National Policy Statement on Indigenous Biodiversity) and as such they are not considered here. Yao et al. (2021) noted that while quantifying ecosystem services in planted forests has recently helped forest companies comply with product certification requirements of the Forest Stewardship Council (FSC), few companies have received a price premium for their certified products.

3.4.2 Baseline water quality footprint

It is challenging to assess the environmental footprint of forestry due to the range of factors that influence outcomes such as underlying geology, hydrology, previous land use, stand age, and forest management and harvesting practices. Nitrogen losses can be separated out across key phases in the forest cycle for farm, plantation and permanent forest. Sediment losses can be separated out across key phases in the forest cycle for farm and plantation forest. However, P losses are best averaged across the forestry cycle at this stage, due to a lack of studies as opposed to consistent P loss across the forestry cycle. In terms of contaminant footprint it is assumed that farm and plantation forestry have the same base footprint, but they are separated out in terms of profit. There is not enough information to include the impacts of one-off weather-related events.

There is information available to separate out base N loss by key stages in the forestry cycle (based on studies such as Davis, 2014). The data represented here excludes differences based on prior land use due to lack of evidence. It is assumed that the base N loss is the same for farm and plantation forest, it is higher for permanent forest due to lower nutrient requirement for permanent trees. These results are summarised in Table 25. There is some literature available on N loss from different soil types; however, with the limited profit and mitigation information there is minimal point in separating out the base N loss by soil type.

Table 25: Base water quality impact estimates: N (kg N/ha/yr)

Years	Pre-plant	1-14	15-27	Post-harvest (approx. 2 yrs)	Average
Farm plantation forest	8	1	4	5	2.5
Plantation forest	8	1	4	5	2.5
Permanent forest	8	1	4 *ongoing	NA	3

There are extremely limited quantified P loss estimates from forestry in the literature, especially across the rotation. Three estimates presented here (Table 26) for the average P loss are the same for farm and plantation forest and lower for permanent forest on account of the permanent forestry cover. These estimates are based on Ledgard (2014), Matheson (2018) and Monaghan (2007). It is noted that Payn and Clinton (2005) had a much higher P loss per hectare (an average of 1 kg P/ha/yr).

Table 26: Base water quality impact estimates: P (kg P/ha/yr)

Years	Average
Farm plantation forest	0.2
Plantation forest	0.2
Permanent forest	0.1

The potential sediment loss base footprints are summarised in Table 27. There is some information on sediment losses from forestry. However, these are often only parts of the forestry cycle, and commonly take the form of sediment yields, not sediment loss to water. Some data is available across key stages of the forestry cycle although, given the uncertainty in this, averages are provided as well. There is likely a difference based on slope. However, there is not enough detail to separate out these results at this stage. There are a large range of sediment loss estimates even across relatively few studies and given the impact of weather it is hard to find a robust figure to transfer to the FWMT. The results for the permanent forest are based on Baillie and Neary (2015). The average sediment loss for plantation and farm plantation forest is based on a mature forest not at harvest time and therefore will likely underestimate sediment loss to water over a total rotation. There is not enough data to differentiate indigenous versus plantation forestry, and it is likely the summary here is wrong in having the same result for permanent and plantation forest, based on the qualitative discussions in literature and the impact harvest and associated earthworks (e.g., roading) has on sediment loss.

Table 27: Base water quality impact estimates: Sediment (kg/ha/yr)

Years	Mature trees (average)	Harvest (average)	Post-harvest (average)	Full rotation (average)
Farm plantation forest	235	575	1,800	875
Plantation forest	235	575	1,800	875
Permanent forest	NA	NA	NA	875

4 Mitigations

4.1 Introduction

This section reviews the literature for potential device and source control mitigation options for forestry. As discussed in Section 3.3, management activities have the potential to affect water quality e.g., Binkley et al. (1999) and Neary (2012). There is some literature on practices that can be undertaken within plantation forestry to reduce the impact of forestry, particularly harvesting, on water quality (Bloomberg et al., 2011). These include actions such as management of riparian areas, changing species types, staggered harvesting, planting density and harvesting practices. However, while these are often suggested as having positive environmental impacts these are often not quantified, nor are the associated economic impacts.

As with the other rural sectors, the quantified economic and environmental impact of mitigation options depends based on the underlying typologies of the forestry action in question, for example, the underlying biophysical characteristics and management of the forest. So, where information on the quantified economic and environmental impact of mitigation options does exist it is often hard to translate from a study site to another location or context. In addition, while information does exist on forests as a mitigation from other land uses, there is less information on key forestry activities, like harvesting, and even fewer comparing types of harvesting, let alone comparing types of mitigations across different forest characteristics (e.g., first versus second rotation) and most studies are dated (i.e., Baillie et al. 2015; Collier and Winterbourn 1989; Jackson 1987; Neary and Leonard 1978). This lack of robust quantified information makes it challenging to estimate the impact of actions to mitigate the water quality impact of forestry and particular forestry actions.

In addition to the lack of general information, there is a potential bias in the information that is available as most studies have focused on particular types of forestry. For example, studies looking at harvesting have predominantly focused on harvesting in steeper hill country (i.e., Baillie et al. 2005; Boothroyd et al. 2004; Eyles and Fahey 2006; Graynoth 1979; Quinn et al. 2004; Thompson et al. 2009) and less so on the areas of flat to rolling topography (i.e., Collier and Bowman 2003; Pruden et al. 1990). Consequently, the published information on the extent of environmental impacts of harvesting on waterways in New Zealand's planted forests is more representative of forestry in steep hill country.

Documents such as the NES-PF consider some conditions that could be considered mitigations for plantation forestry, such as setbacks from waterbodies and installing and maintaining storm water and sediment control measures. However, limited quantifiable detail was provided on the costs and benefits of specific activities.

In addition, estimating the cost and benefit of mitigating the impact of forestry on water quality is particularly challenging given the substantial contaminant loads generated by 'high impact events' during the window of vulnerability. Urlich (2015) noted that the generation of fine sediment associated with forestry harvesting is inevitable no matter how many, and how stringent, the controls due in part to the periodic occurrence of high intensity rainfall events. This is further complicated by the potential for climate change to increase in the frequency of extreme weather events with greatest risks associated with harvesting of planted forests in highly erodible and steep topography (Dunningham et al. 2012). Extreme weather events over large spatial scales can override the effects of land use and mitigation activities (Basher et al. 2011). The window of vulnerability can be managed in part by identifying areas of high risk and taking appropriate action such as retirement, choosing longer-rotation species, increasing stocking rates, planned reversion, and retaining larger buffers (to act as slash traps) between harvest/production areas and streams are potential ways of managing this risk. Understanding and management of the risk associated with high intensity, low frequency storms and their local nature has received less attention (Phillips et al., 2012) and represents a further information challenge.

As landslides and debris flows are natural processes, it is not feasible to stop them completely (SCION, 2017b). Currently, science is not yet able to determine the storm threshold conditions for which these events could be managed, which makes for a challenge when modelling forestry mitigations.

Baillie and Neary (2015) found a number of specific knowledge gaps relating to research in forestry environmental impacts, including that research is lacking on the effectiveness of the latest best management practices, technologies, rules and regulations, and regional and national policies on mitigating the effects of management activities (particularly harvesting) on water quality in planted forest waterways. They also found that there are very few studies on the effects of management activities on freshwaters during the establishment and growing phase of the forestry cycle are limited in scope and dated.

4.2 Mitigation actions

This section considers both device and source control mitigation options. Namely, harvesting methods, post-harvest vegetation management, fertiliser use and land use change are considered as source controls. Sediment control measures, riparian buffers and wetlands are considered as devices. All of these options are focused on plantation forestry (and farm plantation forestry, given this is plantation and is assumed to have the same environmental footprint). They do not apply to permanent forestry which is not actively managed.

4.2.1 Harvesting methods

While forestry is considered to reduce nutrient loss and provide erosion control benefits relative to alternative land uses (such as pastoral farming), harvesting can still have considerable negative impacts, especially on sediment and phosphorus losses, particularly if managed poorly. Options to help reduce the negative impacts of harvesting include selective harvesting, progressive harvesting and alternative harvesting methods. In addition to these options, the negative impacts of harvesting can be (in part) mitigated through the use of riparian areas (see section 4.2.5), post-harvest management (see section 4.2.2) and best practice sediment control (see section 4.2.3). Harvesting is not the only influence on water quality, with associated earthworks also a significant risk. For example, roads and landings are also key risk areas for erosion and increased sediment and phosphorus loss these practices are considered in some studies as part of harvesting and in some as part of sediment control measures.

With trees grown for timber, tree harvesting, particularly under clearfell harvest regimes, can result in sediment and phosphorous loss with associated impacts on waterways (Aust & Blinn 2004; Baillie et al. 2005; Campbell & Doeg 1989; Pike et al. 2010). Sediment yield from forestry blocks increases in the few years after harvesting but then drops to pre-harvest levels even in the absence of large storms (Fahey et al. 2003; Phillips et al. 2005; Marden et al. 2006). While harvesting plantation forestry causes a rapid peak in sediment generation, the use of 'good practice' during harvesting can help sediment loss return to preharvest levels within 1-2 years (Ritchie, 2011). The time taken for freshwater environments to recover to their pre-harvest state varies spatially and temporally across a wide range of water quality attributes (Baillie & Neary, 2015). When forests are harvested, landsliding risk increases. Whether that risk is fully realised is usually related to the incidence of storms in the years following forest removal (Phillips et al., 2015).

An alternative to clearfell harvest is selective tree harvesting under continuous cover harvest regimes, or phasing planting to ensure staggered harvest windows in sensitive landscapes. The former is more likely to be used with non-radiata timber species for high value timbers. Progressive harvesting is spreading harvest over several years or developing a 'mosaic' of harvested blocks across a landscape. This helps reduces the risk of large sediment inputs to waterways due to mid-slope failures during storm events (Ritchie, 2011) and will also limit nutrient run-off and leaching (Knowles & Hansen, 2004). The use of selective harvesting and progressive harvesting have in some cases (along with other mitigations such as retention of riparian buffers) effectively mitigated the impacts of harvesting on water quality (Aust & Blinn 2004; Baillie & Neary, 2015; Broadmeadow & Nisbet 2004; Kreutzweiser and Capell 2001; Little et al., 2014; Neary et al., 2010; Pike et al., 2010). Managing the window of vulnerability is particularly problematic in a clear-fell system, as opposed to coupe harvesting (smaller clusters of trees), selective harvesting or progressive harvesting (Urlich, 2015). This is because greater amounts of sediment are produced in a clear-fell system, due to the area of bare soil exposed and there is no buffering effect from surrounding trees being left which can contain sediment runoff (Urlich, 2015).

The physical characteristics of the forest and forest location will impact on how effective any mitigations are. For example, Environment Southland considered modelling staged/progressive harvest as a mitigation option for contaminant loss from forestry. They noted that while staged harvesting, and continuous canopy harvesting have lower rates of contaminant losses than clear-fell harvesting, staged harvesting was not feasible in Southland due to high winds and the risk of winds causing remaining trees to fall (Moran et al., 2019). This highlights that mitigation options are not always applicable in every context, making generalisations about costs and benefits challenging.

The effect of alternative harvesting techniques (or alternative logging systems) is likely to have a minor direct impact on slope stability compared with the more significant effects of clear-cutting and earthworks associated with road and landing construction (Phillips at al., 2015). However, systems that require more slope disturbance by way of more earthworks, tracking, etc. are likely to increase the potential for slope instability (Phillips at al., 2015). These alternative logging systems have been used to help reduce erosion, and most steep land is now cable-harvested instead of ground-based thus reducing the amount of roads and tracks and slope hydrology disruption (Phillips at al., 2015).

One study (O'Loughlin, Rowe and Pearce, 1980) provides detailed information on two alternative logging systems, downhill cable logging, and a skidder logging technique, based on a study in 1974 in Westland. O'Loughlin et al. (1980) investigated the hydrological consequences of harvesting mixed beech/podocarp/hardwood forest and establishing a *Pinus radiata* crop on steep hill country. Before clearfelling the main experimental basins, two alternative harvesting methods were assessed in two small basins. Without any pre-treatment monitoring, one basin (M7) was clearfelled and harvested by a downhill cable logging technique and the second basin (M9) was clearfelled and harvested by a skidder logging technique.

In O'Loughlin et al. (1980), basin M7 was completely clear-felled and then logged with a northbend skyline system. All logs were dragged in the downhill direction to a log landing near the basin mouth. During extraction dragging logs caused considerable disturbance to slopes and streambanks. Streamside vegetation was entirely removed. There was no tracking on the basin. Basin M9 was logged with rubber-tyred articulated skidders. An access track 1,800 m in length was constructed around the near perimeter of the basin and logs were extracted in the downhill direction to a log landing near the basin mouth. A riparian protection zone of intact forest nominally 20 m wide on each side of the stream and occupying about 2 ha was left. Both basins were burnt after harvest (excluding the riparian area in Basin M9).

In Basin M7 (which was clearfelled and harvested by a downhill cable system with no tracking), sediment yields were not significantly different from the control basin. In Basin M9 (which was tracked, harvested by rubber-tyred skidders and burnt) sediment yield rates increased to eight times the yield rate from a nearby forested control basin, with most sediment derived from the track (O'Loughlin et al., 1980). While the logging system had an impact in this study, it is possible that the largest impact was indirect and associated with the addition of tracks needed for the skidder logging technique. These results are presented in Table 28. The results do note that study years were 'dry' years and it is noted that if study years had been normal rainfall the sediment yields would have likely been higher.

Table 28: Sediment yields from different logging systems in north Westland (Basin M7=downhill cable system, M9=skidder logging system, Basin M6= control), adapted from O'Loughlin et al. (1980).

Land use	Basin M6	Basin M7	Basin M9
Area (ha)	1.63	4.14	8.26
Study period (days)	730	911	545
Suspended sediment load (m^3)	0.73	3.82	26.63
Trapped sediment load (m^3)	0.34	1.08	5.89
Total sediment load (m^3)	1.07	4.90	32.52
Sediment yield rate (m^3/km/year)	33	47	264

Burning forestry slash has been used as a post-harvest management tool although not used widely currently. It can, however, result in loss of nutrients from the site through loss to the atmosphere, surface wash or wind-blow, and leaching. In an Australian study where logging slash of *Pinus radiata* was burnt in a high intensity fire, 84% of the fuel load (79 t/ha) was consumed, resulting in losses of 220kg N/ha (72%) and 8kg P/ha/yr (27%) (Flinn et al. 1979). While burning may result in a quick flush of nutrients into streams, it may be no different to the long-term loss when slash is left to decay, as was found in Rowe and Fahey (1991). Alternative slash management such as utilising off-cuts and "waste" for alternative revenue streams e.g., pulp or wood chip, has not yet been the subject of research.

Harvesting can also have an impact on N leaching. Fahey and Jackson (1997) assessed the effects of harvesting of indigenous forest catchments in Nelson. Harvesting increased total N leaching loss 10-fold and this remained 3-5 times higher in logged than control (unlogged) catchments four years after logging and had not returned to control levels by six years after harvest (Fahey and Jackson, 1997). This study did not quantify the values beyond this. At a West Coast site, a combination of logging indigenous forestry and slash burning increased nitrate-N leaching loss by up to four-fold in the first year, and up to 20-fold in the second year (O'Loughlin et al., 1980). Specifically, nitrate-N leaching losses were increased from about 0.5 kg N/ha/yr in control catchments to up to 10.4 kg N/ha/yr in a catchment that had been harvested followed by slash burning. Longer-term data from catchments that had been harvested with slash either burnt or not burnt indicated smaller losses of N (but still elevated from the control), of the order of 1.2-2.7 kg N/ha/yr (compared to control catchment losses of 0.5 kg N/ha/yr), over an 8-year period (Rowe and Fahey 1991).

Davis (2014) adjusted data from N concentrations in Dyck et al. (1981) to N loss/ha (assuming drainage was one third of annual precipitation at 1,400 mm) and noted that while harvesting caused a rapid increase in nitrate-N concentrations, which persisted through to the end of the study (approximately 36 months after logging), harvesting followed by slash burning caused a shorter-lived response. The two treatments caused only a minor increase (circa. 10 kg/ha) in the amount of nitrate-N leached over the 3-year course of the study (Figure 8).

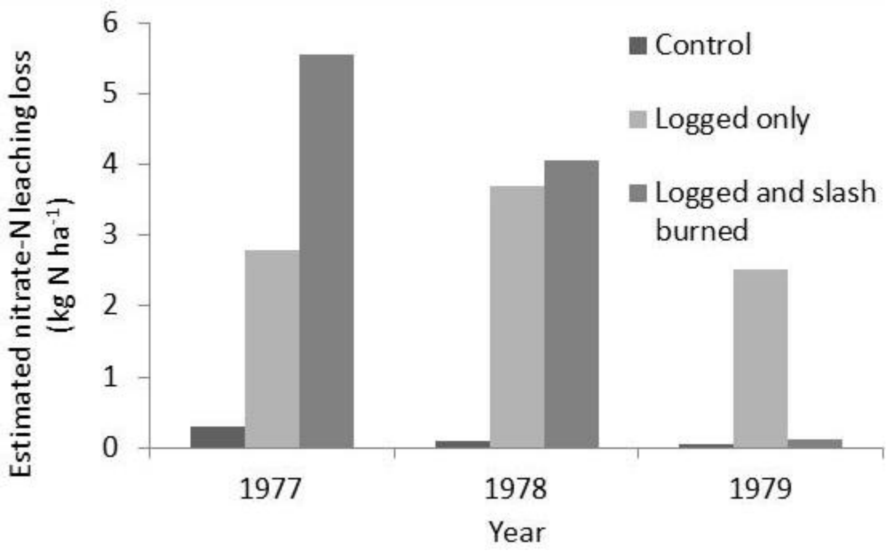

Figure 8: Impact of logging (1976) and burning (1977) operations on subsequently estimated N leaching losses at 1 m depth in *Pinus radiata* forest on volcanic ash (Kaingaroa) (Davis, 2014).

It should be noted that post-harvest nutrient increases do not occur in all cases (Baillie and Neary, 2015). A Hawke's Bay hill-country catchment showed no significant changes in TP, soluble reactive P, total nitrogen (TN), NO_3-N, or NH_4-N (Ammonium-nitrogen) after harvesting (Fahey and Stansfield, 2006). This was possibly attributable to the BMPs employed during the harvest operation (Eyles and Fahey, 2006). In addition, there were no significant increases in NO_3-N after harvest of a small central North Island catchment with a high background concentration of N (Parfitt et al., 2002). It was not clear what exact BMPs were utilised in either of these studies so if these could be attributed as a mitigation in the FWMT or if they are equivalent to the requirements under the NES-PF.

While there are no studies which assess the economic and environmental performance of different harvesting practices (both harvesting techniques and processes) there is evidence that changes in harvesting will have economic impacts. Harvesting costs and equipment are directly related to the slope of the forest being harvested. In harvest planning, slopes greater than 15 degrees are recommended to have equipment suited for steep slopes, for example, harvesting by haulers, while for slopes less than 15 degrees, ground-based equipment is common (Yao et al., 2017).

Economic constraints limit the use of selective/progressive harvesting and small harvesting coupes (Baillie & Neary, 2015). They believe that the greatest gains in water quality will be from advances in harvesting systems and BMPs that eliminate or mitigate harvesting risks, particularly in the steeper forested areas, especially where riparian areas are not feasible (due to slope or harvesting and roading constraints) (Baillie & Neary, 2015).

Existing forest infrastructure can also have a big impact on harvesting costs due to the requirement to build new internal roads, landings and barge terminals for the first rotation (Yao et al., 2017). Conversely the existence of infrastructure can greatly reduce the cost burden and a forest that lost money in its first rotation may become profitable in the second.

4.2.2 Post-harvest vegetation management

Immediately post-harvest is a key time for negative impacts on water quality from forestry as described by the window of vulnerability. Options to mitigate these impacts include managing riparian areas (see section 4.2.5), managing harvest and slash burning (see section 4.2.1) and management of vegetation post-harvest (e.g. managing how quickly and what type of vegetation is established). By leaving a site bare of vegetation, harvesting disrupts nutrient-cycling processes, allowing nutrient leaching to occur. Amounts of nutrients leached are generally small, but may be greater if site revegetation is prevented, or if the soil is high in nitrogen, as on ex-pasture sites (Davis, Xue and Clinton, 2015). Rapid revegetation reduces the potential loss of nutrients after harvest. However, there is limited data to quantify the potential water quality impact or economic cost of this. This section largely refers to revegetation of the next rotation of *Pinus radiata* or allowing weed cover to establish, although there would be the potential to consider alternative vegetative species, this practice could be challenging with stumps and slash remnants.

Where leaching losses occur after harvest, they are usually short lived (likely less than 2 years) since revegetation of cutovers by weeds, which take up nutrients that might be otherwise leached, is rapid (Davis et al., 2014). This highlights that N losses can be higher if revegetation is prevented. Increased concentrations of N were measured in suction cups after above ground vegetation was removed from trenched treeless plots (simulating harvesting) in Kaingaroa Forest (Dyck et al. 1983), who found that:

- After 21 months, soil-water nitrate concentrations had risen to a maximum of 9.4 mg/L (equivalent to 42 kg/ha/yr).
- This was compared with 2.0 mg/L in an adjacent logged but unweeded area.
- Concentrations in unlogged controls were 0.002 mg/L.
- Concentrations in trenched and weeded plots declined to 4 mg/L at the close of the study (24 months after treatment).

SCION (2017b) noted that rapid replanting and replanting at a higher stocking density to shorten the time it takes for the new crop to close canopy, is a practical and easy option to minimise the window of vulnerability. Other options in the long term could be planting coppicing species that maintain live roots after harvest, though this is as yet unproven and there is limited information on the economic impact of this method (SCION, 2017b).

Similarly, Urlich (2015) found that one of the key options to improve improving soil conservation and water quality in the Marlborough Sounds was requiring replanting of *Pinus radiata* to reduce the window of vulnerability, specifically replanting of areas harvested within 12 months of harvest and replanting in excess of 1,000 stems per ha. The potential benefit of this was not quantified though.

Davis et al. (2014) discusses how the presence of weeds can have a beneficial effect on bare sites by minimising nutrient leaching, especially of N; thus “blanket” weed control (such as spraying) should be avoided. Nitrogen leaching losses at harvest may be most effectively reduced by rapid establishment of a vegetation cover after harvesting (Dyck et al., 1983; Parfitt et al., 2003). A cover of ‘weeds’ normally develops rapidly after harvest at most forest sites, however where this doesn’t occur, grasses or other herbaceous species may be introduced by over-sowing (West et al. 1988, West 1995).

Competition removal by “releasing” spots (spraying herbicide to the area around a tree for vegetation control) around trees at planting also causes increased leaching losses. Parfitt et al. (2003a) measured N leaching under released spots, rank pasture between trees, and nearby grazed pasture. The N leaching losses over a 14-month period amounted to 39 kg N/ha under the trees (with herbicide areas), 15 kg N/ha under the rank grass and 3 kg N/ha under the pasture comparison. The herbicide-treated area under the trees was only 12% of the total land area, so the scaled contribution to leaching under the trees amounted to 5 kg N/ha.

Vegetation strategies should consider the underlying N fertility of a site (Davis et al., 2014). A site with a high background N content (sites where a response to additional N would not be expected) should not be replanted by N-fixing species, which may increase N leaching. On high background N sites where N-fixing species (shrub weeds or herbaceous legumes) are likely to naturally occur after harvesting, oversowing with grasses can be used to attempt to reduce establishment of N fixers and the risk of N leaching at the site (Davis et al., 2014). Conversely, on sites with a low background N content, establishment of N-fixers could be beneficial and result in reduced N leaching from the site (Davis et al., 2014). On low-N sites, oversowing with a pasture legume, in conjunction with application of phosphate fertiliser and weed control around tree seedlings, is an option to increase site N fertility without increasing the risk significantly of N leaching site (Davis et al., 2015). As shown in Dyck et al. (1983) N loss under gorse can be as high as 23 kg/ha/yr and rates of 40–60 kg/ha/yr of N loss under gorse have been measured in the Bay of Plenty area (Davis et al., 2014). This reiterates that vegetation management between harvest and into replanting is important until the next rotation of trees is established.

4.2.3 Sediment control measures

Sediment loss is generally one of the biggest environmental issues for forestry. However, mitigation of sediment is challenging as sediment loss occurs in varied rates across the forestry cycle, especially in the window of vulnerability, during key activities, including roading and harvesting and is heavily influenced by weather events, particularly storm events that coincide with key activities and the window of vulnerability. Storm events cannot be controlled, and so mitigation focuses on managing sediment generation and reducing the volume/velocity of storm water run-off to reduce its power to entrain sediment.

Mitigation measures are available for both erosion control and sediment retention (Philips and Marden 2003). Erosion control measures include management of water flows (e.g., with contour drains and culvert flumes) and protection of exposed soil (e.g., by hydroseeding, mulching, riprap, erosion blankets, armoured water tables, brush layering, or surface roughening to encourage revegetation). Sediment control measures aim to prevent any eroded soil from being transported to nearby streams. They include silt fences, diversion channels or bunds, sediment traps, pits or retention ponds, and measures to reducing the hydraulic conductivity between roads and streams. In addition, sediment mitigation measures also include harvest planning, including protecting steep slopes and water courses, road design and decommissioning, retention and enhancement of riparian vegetation and slash barriers and management (Ritchie, 2011). Despite sediment and erosion control measures being critical to minimising the impact from forests into waterways and being a focus of the NES-PF and the NZ Environmental Code of Practice for Plantation Forestry, there is limited quantified information on the relative cost and efficacy of mitigation options.

Forestry operations can expose soil and increase losses of suspended sediment, particularly during afforestation and harvesting. During these times, shrub clearance, loosening of soil and infrastructure works can make the soil vulnerable to erosion from rainfall and wind, and increase the risk of large sediment loads entering water bodies (Ledgard, 2014). During the plantation forestry cycle, earthworks (for roads and landings) and clear-felled areas have the potential to generate large amounts of sediment by both surface erosion processes and mass movement (Basher et al., 2019). The extent of run-off from tracks is affected by the proximity to waterways, slope, stability of cuttings and frequency of use. Designing regular cut-offs into vegetated ground before flows become strong enough to become highly erosive is a key means to reduce sediment from these sources (Quinn et al. 1997). While mechanical slope stabilisation is generally not economically feasible along most low volume roads and tracks, sediment loss can be partly reduced by road location and construction methods that recognise erosion hazards (Phillips et al., 2012).

Forest landings are also a potential source of sediment loss in plantation forests. Once considered a key source of sediment generation through mass failure (e.g., Coker et al., 1990), the location, construction, stabilisation and management of landings are now much improved (Phillips et al., 2012). However, they can still be a key source of sediment loss. Despite this the quantification of cost and benefit of mitigation options in landing construction are not quantified.

Reviews of the effectiveness of forestry best management practices to reduce surface erosion are more prevalent in USA but still focus on demonstrating the effect of implementing multiple best management practices including silvicultural options, road and track management and stream crossing management rather than specific changes. They show that best management practices can minimize erosion and sedimentation of more than 50% but implementation rates and quality are critical to efficacy. None of the reviews specifically mention management practices aimed at minimising landslides or debris flows which are considered the main contributors to erosion and sediment generation in New Zealand (Phillips et al. 2012).

Erosion controls can also retain nutrient pools in the soil, thereby supporting forest productivity. Very limited data exists for the impact of erosion on planted forest soil nutrient pools and impact on forest productivity. One example, at Pakuratahi, shows the loss of soil through shallow landslide erosion to result in a reduced timber volume of 10% for trees planted within the erosion scars (Heaphy et al., 2014). However, given the lack of quantification in relation to erosion effectiveness and cost it is difficult to incorporate the productivity impact.

4.2.4 Fertiliser use

Nitrogen and P losses from forestry are, in general, relatively low compared to other land uses, in part, because of relatively low fertiliser use. Where it is used, fertiliser is generally only applied a few times during a 25-to-30-year rotation. Despite this low fertiliser use, there is potential for changes in fertiliser practices (where used) to reduce N and P loss. There is some literature on the growth rate response to various fertiliser application rates and some on nutrient losses associated with fertiliser use. However, as with other mitigations the evidence base is limited and there is not a lot of published information linking N and P fertiliser use with losses of nutrients to waterways as well as productivity impacts. Due to the complex nature of forest trials and data it is hard to link studies that cover either economic or environmental aspects. Davis et al. (2015) provides a good summary on fertiliser use, fertiliser responses and fertiliser impact on nutrient losses, but these are all drawn from different studies and therefore can't be readily linked to assess the environmental and economic impacts of changing fertiliser use.

Fertiliser use is also complex in forestry as it does not just influence growth rates but also wood quality. It is necessary to understand the impact of fertiliser application on tree form and wood properties as well as wood growth. The wood properties most influenced by fertiliser application in forests are ring width, the proportion of latewood, fibre characteristics and wood density (Davis et al., 2015).

Nutrient losses via leaching will vary from year to year, as they depend on climate (leaching cannot occur without drainage) and site factors. When high rainfall causes drainage, the sites most at risk for nutrient leaching are those with very high/repeated doses of fertiliser and those with coarse textured soils.

The average application rate of all fertilisers used across planted forests in 2017 (including those with no applications) was 8 kg/ha, although actual application rates would be higher (common rates are 200 kg N/ha and 75 kg P/ha (SCION, 2019). The decision to add fertiliser will depend on factors including harvest methods and site fertility. Despite low fertiliser use currently, some areas of New Zealand planted forests are currently entering their third and fourth rotations and fertiliser use may increase in the future to boost productivity and/or maintain soil nutrient sustainability over successive rotations (SCION, 2019).

Urea is the most commonly applied N fertiliser to forests, it is applied at rates up to 450 kg/ha (207 kg N/ha) (Davis et al., 2015). Results from N-rate trials with *Pinus radiata* from around New Zealand indicate that the response generally appears to plateau at about 200 kg N/ha and the normal recommendation is for aerial application of 200 kg N/ha (Will, 1985; Olykan, 2002).

For P, the literature indicates that while growth responses may occur up to very high rates of applied P, most of the response is achieved at rates of 60–70 kg P/ha (Davis et al., 2015). The soil P retention level appears to have little bearing on the rate or type of fertiliser required. Most of these studies are done on clay soils, the only P rates trial on a soil outside of the northern clay and podzol group, on a Brown soil, indicated rates of 40–50 kg P/ha may give optimal growth (Davis et al., 2015).

Base levels of nutrients in the soil have an impact on likely response rates. Where nutrient deficiencies severely limit tree productivity, large responses to fertiliser are possible (Davis et al., 2015). In a series of five trials established at sites where a fertiliser response to N and/or P was considered likely, four-year responses to N and P ranged from -14 to +67% (I. Hunter et al. unpublished, as cited in, Davis et al., 2015 Table 9.5). This study used approximately five-year-old *Pinus radiata* forests and were mostly unthinned and unpruned, the rate of fertiliser was not reported in Davis et al. (2015), this is reproduced in Table 29.

According to I. Hunter et al. (unpublished, as cited in, Davis et al., 2015), small additions of N and P produced a 20–30% increase in the annual growth increment for a podzolised sand in Northland, but larger additions of fertilise reduced the rate of response in the growth rate.

Table 29: Four-year volume and proportional response (% in brackets) to N and P fertiliser by *Pinus radiata*, standing volumes and periodic current annual increments are given for control plots, adapted from I. Hunter et al. unpublished, Table 9.5, in Davis et al. (2015).

Site	Standing volume (m^3/ha)	Periodic current annual increment (m^3/ha)	Volume response to N alone (m^3/ha)	Volume response to P alone (m^3/ha)	Volume response to N & P (m^3/ha)
Podzolised sand Northland	45	10.6	11.8 (28)	13.7 (32)	12.1 (28)
Podzolised clay Northland	159	33.7	-10 (-7.4)	-7.5 (-5.6)	28 (21)
Pumice southern Kaingaroa	83	18.8	-6.3 (-8.9)	-7.1 (-9.4)	3 (4)
Soil from granite Nelson	80	17	-9.7 (-14.3)	36.9 (54.1)	41.7 (61)
Pakahi Westland	69	15.3	1.1 (1.8)	10.5 (14.7)	41.1 (67.2)

According to Davis et al. (2015) there are a number of key factors that influence the level of response to fertiliser (and therefore the potential considerations when reducing fertiliser use to minimise risk of N and P loss to water):

- For N, the size of the response depends on the openness of the canopy.
- The greater the amount of available nutrient in the soil, the smaller the response from additions.
- The amounts of other potentially limiting nutrients in the soil. For example, if N is added but, if P is limiting, there could be a nil or even negative response to N additions.

- The age and silvicultural history of the stand. Unthinned stands will respond provided they have not closed canopy, therefore young stands may respond, and thinned and (or) pruned stands often respond well, especially to N. Older stands at or near canopy closure are unlikely to respond.
- The presence of weeds. Fertiliser application will enhance the growth of competing weeds and can limit the response to fertiliser through competition for light, moisture and other nutrients. Therefore, in young stands where fertiliser is applied prior to canopy closure, it may be necessary to control weeds to obtain benefit from fertiliser.
- Disease. Stands affected by fungi that reduce needle mass, effectively thinning the canopy, may respond to fertiliser, especially N, which will be used to rebuild the canopy.

Binkley et al. (1999) reviewed the results of several dozen studies from around the world on the impacts of applying N and P fertilisers on stream water quality. They concluded that fertiliser application to forests commonly leads to moderate increases in stream water nutrient concentrations. The greatest increases came from direct application to streams, use of ammonium nitrate fertilisers and/or the application of high rates or repeated doses.

Two studies (Leonard 1977; Neary and Leonard, 1978) covering a range of New Zealand sites with varying geology, climate and hydrology found fertiliser application to trees on nutrient-deficient sites resulted in short-term increases in N and P in forest stream waters. The main pulse of N or P was detected either immediately (particularly when application was over the stream channel), just after fertiliser application, or during high flow events in the weeks or months immediately after fertiliser application. In general, nutrient levels returned to baseline levels within a few weeks or months of application. The authors did not consider the levels in stream water to be harmful as only a small portion of the total amount of fertiliser (0.5% of total application) reached the streams. However, since these study periods, fertiliser application systems have improved and it is likely that the pulse of N or P just after application is less likely now, especially as fertiliser is unlikely to be applied over stream channels as per the NZ Environmental Code of Practice for Plantation Forestry (NZFOA, 2015).

There are a few New Zealand-based studies which quantified N losses following fertiliser use. This includes a lysimeter study by Worsnop and Will (1980), on a Taupo silty sand soil in Kaingaroa Forest. This showed no effect of fertilisation (200 kg/ha of urea) on leaching loss of N or other nutrients for the three years of monitoring after fertiliser application. Similarly, Clinton and Mead (1994) were able to recover in soil and plant material all of the 15N-labelled N they added to a young (4-year-old) stand of *Pinus radiata* on a silt loam soil in Canterbury, indicating leaching losses would not have occurred in that environment.

Studies in sand soils indicate that N leaching occurs more readily in these environments. Thomas and Mead (1992) applied 15N-labelled N (150 kg N/ha, broadcast) to 2-year-old seedlings on bare coastal sand in Canterbury in either single or split applications. For the single application, 30% was lost below the main rooting zone through leaching, but when split into three applications of 50 kg/ha, more N was retained in the soil and none was detected below the root zone at 80 cm depth, irrespective of whether the fertiliser was applied in autumn, spring or summer. However, it should be noted that splitting applications will incur more labour and machinery costs than if fertiliser was applied in a single application.

At Woodhill Forest in Auckland, Smith et al. (1994) compared N leaching (in suction cups at 60-cm depth) following harvest of a 42-year-old stand, in the presence and absence of urea application. Where urea was added (200 kg/ha/yr in quarterly additions of 50 kg/ha for 2 years), N concentrations had increased in the leachate 20 weeks after harvest and remained at elevated levels in most treatments throughout the monitoring period to nearly 2.5 years after harvest.

The general best practice rules for fertiliser use apply equally for forests as well as pasture, but there is generally no quantification of these for forestry. For forestry, these best practices include avoiding fertiliser use when soil moisture is too high to cause leaching or runoff, applying split applications where suitable to avoid high rates and avoiding applying fertiliser where it could enter waterways (Davis et al., 2015). Specifically for N, Davis (2005) proposed a conservative approach to prevent leaching of N and suggested the following guidelines for fertiliser applications in forestry:

- Broadcast applications of N at any time should not exceed 200 kg/ha.
- Spot applications after planting should not exceed 30 g N/tree.
- Application during winter or other wet periods when soil water drainage is likely should be avoided.
- Except where planted forests are severely deficient, broadcast applications should be made in conjunction with thinning and pruning operations when N demand is high.
- Application decisions should be based on results of nutrient analysis of foliage.

However, as above, while these provide practical guidelines there is no robust quantification of the impact of these on nutrient loss or economic performance across landscapes.

4.2.5 Riparian buffers

Maintaining riparian buffer areas alongside waterways has been considered an option to mitigate the impact of forestry on waterways, particularly the effects of harvesting (Baillie and Neary, 2015). Riparian buffers have been the focus of much of the literature concerning estimated costs and benefits of those mitigations available to plantation forestry. This traditional focus is possibly because the cost estimate is relatively straightforward (as it is primarily based on area removed from production), though there is less consensus on potential benefits. As with pastoral research, there are no consistent, comparable studies to evaluate the relative benefits of differing riparian widths for all potential contaminants.

Riparian areas (or setbacks as referred to in the NES-PF) were included as a requirement in the NES-PF and the potential costs of these were estimated by NZIER (2016). In this work the cost was based on an opportunity cost of not planting setbacks and was estimated at $8,500 /ha for slopes under a 7% gradient and $5,000 /ha for slopes between 7% and 15%. Slopes over 15% were not valued as it was expected that under the status quo these would have setbacks of at least 10 m to reflect forestry practicalities. No estimates of effectiveness were provided. These estimates could be revised for a specific context based on expected annualised income.

One study employed expert opinion to estimate the potential benefits of setbacks on sedimentation caused by plantation forestry (NIWA, 2010). This Waikato-based study used expert elicitation to estimate the benefits of setbacks and gross margin analysis to estimate the economic impact. NIWA (2010) looked at 5 m wide setbacks and assumed there was 60 meters of stream bank per hectare, half of the setback width was assumed to be harvestable area. They estimated a 20% reduction in sediment yields, a 10% reduction in N yields and a 15% reduction in P yields as well as benefits through stream shading and habitat protection. While this is not entirely clear, it is assumed that this is a reduction across the modelled forestry ‘farm’. Economic impacts were a 3% reduction in harvestable area and a gross margin of $773/ha/yr was used (26-year time-period with a stumpage value of approximately $27,000 per hectare).

Lakel et al. 2010 found that stream management zones (riparian areas) ranging from 7.6 m to 30.5 m in width (16 sites) were generally effective in trapping sediment and found no significant differences in sediment trapping ability across the range of widths. Quinn and Halliday (1999) studied large-scale forestry harvest operations at Tairua and found that in a catchment where a 20 m riparian buffer of mature redwoods was left, no changes were observed in stream temperature, water clarity, sediment particle size or periphyton biomass following harvesting. By contrast, in a nearby catchment where no riparian buffer was left after harvesting, there were marked changes in temperature and sediment levels following intense rain (Quinn and Halliday (1999). Boothroyd et al. (2004) found that bank erosion was significantly higher in harvested sites without riparian buffers than those with buffers, even when buffers had a high proportion of grass rather than native vegetation cover.

While riparian areas can reduce the amount of harvesting debris deposited in the stream and contribute to bank stability, they are limited in their ability to filter out point source sediment (e.g., from roading and earthwork construction, e.g. skid sites) and any associated sediment-bound phosphorus (SCION, 2017a). Riparian areas can also limit nitrate levels following harvest, helping to maintain the aquatic community at similar levels to those found in mature pine and indigenous forest streams.

Urlich (2015) found that one of the key options to improve soil conservation and water quality in the Marlborough Sounds was planted (planted in radiata pine but not harvested) setbacks from the shoreline, though the expected benefit was not quantified, nor was the cost. A study which linked to Urlich (2015) was Yao et al. (2017). They found that coastal setbacks (not from streams) of 30 m, 100 m and 200 m in the Marlborough Sounds will reduce the harvestable area, log volume and increase the cost of harvesting (due to volume of logs available to be harvested at any particular extraction point), all contributing to a decrease in revenue from forestry (around 16% for a 200 m setback). But that increased setbacks could avoid more sediment loss (6% more avoided sedimentation from a 200m setback) the FIF was used to generate these economic and environmental impacts. They also made a note that the NPS-PF regulation allows all currently planted forests to be harvested, therefore all benefits and costs discussed will not be realised until the end of the next rotation subsequent to the setback being put in place (Yao et al., 2017). The results from Yao et al. (2017) are summarised in Table 30.

Table 30: Summarised results from Yao et al. (2017) for costal setback scenarios in plantation forestry in the Marlborough Sounds**.**

Setback scenario	Proportion reduction in modelled forest returns	Change in FTEs/year (% in brackets)	Avoided sediment index (no setback is reference)
None			100%
30 metres	1.3%	1.1 (1.2%)	100.3%
100 metres	5.8%	5.7 (6.5%)	102.1%
200 metres	15.9%	14.8 (16.8)	105.7%

Not all studies on riparian buffers find a positive relationship with sediment reduction. However, Graynoth (1979) found that the capacity of riparian buffers to filter out sediment from point-source pollution was limited. For example, while a 30- 150 m wide buffer at a harvested site in Nelson was effective in maintaining substrate composition, it was ineffective in excluding point-source sediment from a skid site or finer suspended sediment (from point-source routing of runoff along skid trails) from the stream channel (Graynoth, 1979). Fine sediment inputs also increased at a Southland site with a 10 m riparian buffer (Thompson et al., 2009).

While a number of studies have evaluated the effectiveness of riparian buffers in mitigating harvesting disturbances in New Zealand, information on the appropriateness and effectiveness of different buffer widths is lacking (Richardson et al. 2012). In particular, the effectiveness of the 5 m setback in mitigating harvest impacts has never been fully evaluated (Baillie & Neary, 2015).

Riparian areas can also work to reduce nutrient losses. Thompson et al., (2009) found in a 10 m riparian buffer helped reduce the post-harvest pulses in N and dissolved reactive P increases relative to a site with no riparian buffer. This study was based in Otago/Southland and assessed the effectiveness of a 10 m buffer in protecting waterways at one site, which was hauler logged with full suspension of logs off the ground (Thompson et al., 2009).

There are some challenges with riparian areas in forestry. For example, while the retention of riparian buffers can mitigate many of the effects of clear-cut harvesting alongside New Zealand streams, a legacy of planting along stream margins means that harvesting up to the stream edge still occurs throughout New Zealand's planted forests (Baillie and Neary, 2015), as the NES-PF allows for the harvest of previously planted forests (prior to its introduction in 2018). In addition, retaining an intact riparian buffer during harvesting remains a challenge in some of the steeper, less accessible areas of planted forests in New Zealand, where roading constraints necessitate the extraction of timber across headwater catchments.

Although the NES-PF already assumes that 5 m riparian areas around waterways are required, meaning the opportunity for riparian buffers for productions forestry in the FWMT is limited to buffers greater than 5 m in width (noting that the NES-PF already restricts afforestation within 10 m of a perennial river with a bankfill channel width of greater than 3 m). The FWMT would be strengthened if the opportunity for riparian areas is clarified (i.e., assess if 5 m riparian areas are applied in all forestry areas in the region as well as how many meters of waterways are in a hectare of plantation forestry on average as most profit estimates are on a per hectare basis). In addition, it appears from the literature that there is likely to be reduced performance of riparian areas on slopes over 15 degrees and therefore this should be considered in future refinements of the opportunity and effectiveness for riparian buffers.

4.2.6 Wetlands

Mitigations such as wetlands (natural or constructed) are applicable to minimise the environmental impacts of forestry, as they are for pastoral land uses. These mitigations can be costed based on assumptions such as size, type of construction and forgone production, although there is a lack of information on the costs of construction in a forestry context.

The quantum of benefits will also vary based on catchment characteristics. However, there is limited research into the benefit of wetlands in forestry areas. Published information is lacking on wetlands and their influence of forest management activities on water quality (Baillie and Neary, 2015).

4.2.7 Land use change

One option to mitigate the impact of forests on water quality is to change land use. For example, removing harvesting as a risk factor would mean changing from production land use to permanent forest cover. This would mean the removal of production income (from timber) but would (currently) retain carbon income, which based on the assumptions in section 3, is more profitable than plantation forestry, although it removes the option for this land to be harvested in the future and this relationship may change under future prices for carbon and regulations (e.g., the change to the 'average accounting' process under Emission Trading Scheme and the potential change in the eligibility of exotic forests for "permanent" sequestration). In addition, some costs may be required to support the transition from plantation forestry to permanent forestry which are not considered here, i.e., the 'cost' implication is the change in profitability and no capital costs, ongoing maintenance costs, or transition costs are included. Consideration needs to be given to the practical implications of these land use changes, for example, plantation and permanent forest cannot transition to farm plantation forestry, plantation can transfer to permanent forest and it is unlikely that permanent forest transitions to plantation forestry.

Forests utilise N in the soil most early in the life of the forest during canopy development. At some time after canopy closure, when there is less demand on soil N from the tree crop, N leaching increases. A land use change to using fast growing short rotation crops such as *Eucalyptus* for wood chip or pulp production could be considered in areas where there is a strong need to reduce N leaching. However, this does impact the profitability of plantation forest and it could increase the sediment and phosphorus loss associated with harvesting.

4.3 Summary

This section reviewed the literature for potential device and source control mitigation options for forestry. There are two aspects to be considered as part of this assessment, the economic and environmental impact of the mitigation and the opportunity that exists to apply these mitigation options.

Ideally identifying the opportunity would be based on an understanding of current practices across the forestry landscape. However, there is a lack of data on the current range of practices across the forestry landscape in the Auckland region and there is also a lack of research on the different base performance and environmental impacts of mitigation options across different practices. It is recommended that more understanding of the current state of practices is sought before deciding on the opportunity for any mitigations for forestry. It is suggested (see section2.6) that the NES-PF should be considered as the first mitigation bundle, essentially as good management practice. However, there still needs to be an estimation of the opportunity for this mitigation to be applied.

The greatest limitation affecting modelling mitigation choices for plantation and farm plantation forest is the absence of quantified research comparing opportunity and outcomes of mitigations on forested land (i.e., Baillie et al. 2015; Collier and Winterbourn 1989; Jackson 1987; Neary and Leonard 1978). This lack of robust quantified information makes it challenging to estimate the impact of actions whether represented as a source control or device in the FWMT, for contaminant and hydrological processes. As such, the following mitigations are described in the literature but cannot be robustly quantified for inclusion in the FWMT at this stage:

- **Harvesting methods** – While O'Loughlin at al. (1980) showed a large difference in sediment yields between downhill cable and skidder logging systems, it is an older study and based in one location (with relatively high rainfall to other areas of New Zealand). In addition, there is no comparable economic assessment of different methods. Likewise, there are some older assessments on slash burning, but economic impacts are not quantified, and this practice is not widely used.
- **Post-harvest vegetation management** - Parfitt et al. (2003a) demonstrated that herbicide use can increase N loss, as it minimises/slows vegetation growth and thereby N uptake, however there is paired economic costs and benefits of this practice. Likewise, Davis et al. (2014) discussed post-harvest vegetation management and its impact on N loss, but the economics of this practice weren't quantified. In addition, the environmental benefits are very case-specific and challenging to translate to an average forestry model in the FWMT.

- **Sediment control measures** – While sediment control measures are discussed regularly, they are not quantified as they are very site specific. In addition, it is thought some sediment control measures will be captured in the NES-PF good management practices. There is no robust data to use to estimate the impact of this mitigation option.
- **Fertiliser use** – There is some literature on the growth rate response to various fertiliser application rates and some on nutrient losses associated with fertiliser use. However, as with other mitigations the evidence base is limited and there is not a lot of published information linking N and P fertiliser use with losses of nutrients to waterways as well as productivity impacts. Fertiliser use is also complex in forestry as it does not just influence growth rates but also wood quality. The average application rate of all fertilisers used across planted forests in 2017 (including those with no applications) was 8 kg/ha (SCION, 2019) meaning there is minimal scope to reduce this further. Studies that were done previously did not utilise current best practice such as avoiding waterways (in the NZCOP), meaning the potential benefit of going beyond current practices will have reduced.
- **Wetlands** – While these mitigations can be costed based on assumptions such as size, type of construction and forgone production, there is a lack of information on the costs of construction in a forestry context and the pastoral costs are likely unrealistic. In addition, the quantum of benefits will vary based on catchment characteristics. Published information is lacking on wetlands and their influence of forest management activities on water quality (Baillie and Neary, 2015).
- **Land use change** – There is not enough information to consider land use change between tree species. In addition, the tree types could have both positive and negative impacts across different contaminants, for example, more rapidly growing trees will likely lead to more sediment due to increased harvesting, but potentially less N loss as more rapidly growing trees increase N uptake. There is the potential to swap between the farm plantation forestry and plantation forestry to permanent forestry; however due to the scarce literature there would be no difference in sediment yield, and the difference in N and P loss is minimal.

The following mitigation option is described in the literature in enough detail that it can be quantified for inclusion in the FWMT at this stage, although there is still high uncertainty based on the available information:

- **Riparian areas** are a device control option for plantation (and farm) forestry. The NES-PF requires 5 m riparian areas around waterways, meaning the opportunity for additional (i.e., above existing footprint) riparian areas in this version of the FWMT is to increase them to greater than 5 m (noting that the NES-PF already restricts afforestation within 10 m of a perennial river with a bankfull channel width of greater than 3 m). Riparian areas only apply to farm and plantation forest as they are designed to help minimise impact of forest harvesting (and the associated window of vulnerability).

  There is some data which could be used to quantify the economic and environmental impact of riparian areas in the literature, however the estimates do vary significantly. The cost estimate is relatively straight forward (as it is primarily based on area removed from production). It is recommended that the annual profit (on a per hectare basis) is reduced by the area taken out for riparian areas. This would require further analysis by Auckland Council to determine the area removed from production.

  There is less consensus on potential benefits, especially of the relative benefits of differing riparian widths for all potential contaminants. Lakel et al. 2010 found that stream management zones (riparian areas) ranging from 7.6 m to 30.5 m in width (16 sites) were effective in trapping sediment but found no significant differences in sediment trapping ability across the range of widths. NIWA (2010) estimated a 20% reduction in sediment yields, a 10% reduction in N yields and a 15% reduction in P yields for a 5 m buffer with a stream density of 60 m of stream per hectare across the modelled forestry 'farm'. The estimates from NIWA (2010) could be applied to 10 m buffers applied to waterways in plantation and farm plantation forestry during harvest and early-stage growth (through harvest and the window of vulnerability). However, care should be taken as with the introduction of the NES-PF, foresters already have a 5 m buffer, the evidence in Lakel et al. (2010) might mean there is limited additional benefit from moving to a 10 m buffer. While the NIWA (2010) estimates could be utilised for 10 m buffers at this stage of FWMT development, this area should be further investigated in future FWMT iterations, if possible, and could be informed by specific forestry modelling such as the FIF.

Clarification of the opportunity for riparian buffer areas would strengthen the FWMT. Assessing if 5 m riparian buffers are applied in all forestry areas in the region as well as how many meters of waterways are in a hectare of plantation forestry on average is needed, as most profit estimates are presented on a per hectare basis. In addition, it appears in the literature that there is likely to be reduced performance of riparian buffers on slopes over 15 degrees and therefore this should be considered in future refinements of the opportunity for riparian buffers.

5 Recommendations for FWMT

5.1 General

Forestry is challenging to incorporate into the FWMT. The two main reasons for this are the lack of publicly available research and the variation in economic and environmental impacts across the production forestry cycle. The lack of research means that the economic and environmental impact of mitigating environmental impacts of forestry across various typologies has not been sufficiently quantified and will be far more challenging to incorporate than either for pastoral or horticultural HRUs within the FWMT. Should mitigation options (both source control and device options) for plantation forestry be incorporated into the FWMT's cost-optimised, dynamic intervention routines, as they are for horticultural and pastoral HRUs, a clear incomparable confidence in model outputs would exist.

The standard use of NPVs for plantation forest cost-modelling aligns well with the LCC framework in the FWMT. However, the 50-year LCC timespan is challenging as it does not align with a typical *Pinus radiata* rotation. While this can be overcome, in part, through the use of an equivalent annualised annuity at this stage of the FWMT, in future iterations, it is suggested that specific forestry modelling is undertaken to capture economic impacts across the forestry cycle as well as assess the best way to incorporate multiple 27-year plantation rotations into the 50-year LCC.

There are a wide range of forest ages across the Auckland region and forests in different rotations (e.g., first rotation, second rotation forests). Forest areas that are in different rotations are likely to have differences in costs and environmental impacts (i.e., first rotation forests often have higher costs and environmental impacts as forest infrastructure such as tracks need to be created). Ideally, these would be treated as different typologies in the FWMT. However, there is not enough data to robustly quantify the differences in these costs and benefits (i.e., to map existing let alone baseline first harvest rotation forest). Likewise, soil and slope differences, key components for HRU development, are not able to be recommended for either base footprints or mitigation performance at this stage of the FWMT development. It is recommended that the ESC, or similar data on geology and slope, is further assessed as a potential delineation in forestry groups to better estimate the base footprints and the mitigation effectiveness. However, this needs to be considered alongside the resolution of available information on mitigations.

Ideally, the economic and environmental footprints (at base state and after mitigations) would be representative of key stages in the forestry cycle, namely early-stage growth, mature plantation forest and harvested forest/sapling stage. The benefit of representing the three ages separately, is the ability to better represent event-based contaminant loads from plantation forest which differ across the forest cycle (e.g., the disproportionate contribution of plantation forest within the window of vulnerability). In addition, separating the production forestry cycle into distinct stages would allow for the more accurate representation of mitigations (e.g., good management practices) that typically target the window of vulnerability due to the increased opportunity of mitigations to treat production forestry land in this stage of the cycle and the greater risk of negative environmental impacts exhibited by harvested forest land. Such a recommendation might be feasible only in Stage 2 of the FWMT development given near-completion of Stage 1 cost-optimisation and dynamic intervention modelling. Implementing this recommendation would require further modelling and analysis.

Based on these challenges and the available information on the economic and environmental base performance, as well as the mitigation performance, of production forestry in the FWMT, it is recommended to represent these components as annualised average values. However, it must also be noted that weather events occurring within the window of vulnerability are likely to cause significant environmental impacts that are not captured in an averaging approach. These events will have an impact; however, the available data precludes the capture of these aspects within the FWMT which, by design, like all models, is a simplification of reality.

Ideally, defining the opportunity for mitigations would be based on an understanding of current practices across the forestry landscape. It is recommended that more understanding of the current state of practices is sought before deciding on the opportunity for any mitigations for forestry. It is suggested that the NES-PF should be considered as the first mitigation bundle, essentially as good management practice. However, there still needs to be an estimation of the opportunity for this, and for further mitigations to be applied.

It is recommended that for future iterations of the FWMT specific modelling of forestry scenarios will improve the profitability estimates for both the base footprint and mitigation options. The environmental impacts could also be reviewed by technical experts especially in relation to their relativity across key phases of the forestry cycle and dependences on differing slope and soil groups. Expert opinion may also be able to provide quantifiable impacts of source controls and further devices that could be applied as mitigation options.

It is recommended that for future iterations of the FWMT the forestry groupings are redefined. Initially this should be based on farm plantation forestry, permanent forestry and plantation forest. Following this, the plantation forest and farm plantation forestry groupings could be segregated based on first rotation versus second (and subsequent) rotations, if there is profit and environmental data available that can also match these segregations. All forestry groups could then be further segregated based on geological and slope factors (rather than just soil type). However, as with the different rotations, these sub-groupings need to be able to have profit and environmental data differentiated for each sub-group or there is limited benefit in separating these groups.

5.2 Baseline profit

As discussed in Section 3.2, profitability of forestry depends on what is included in the returns, and the primary income streams at the moment are timber and carbon. Throughout the research there is a wide range of profitability, different metrics are used, and it is difficult to separate out the NPVs in research into component parts. Using a model, such as FIF, to generate profitability metrics for the different typologies that are specific to the Auckland region and are underpinned by discussions with foresters in Auckland is desirable, this would also enable profitability to be separated out into key forestry phases to align with the environmental impacts across the production forest cycle (both for plantation forestry and on-farm plantation forestry). However, in lieu of this, the following profitability estimates are recommended (Table 31).

Table 31: Annualised profitability estimates

HRU	Annual profit (nominal $/ha/yr)
Farm forest	$185
Plantation forest	$430
Permanent forest	$500

It is worth noting that the permanent forestry profitability is closely linked to carbon prices, and these could change rapidly in the future and throughout a 50-year LCC. As such, this should be considered as part of a sensitivity analysis when assessing profitability of forestry in any further modelling.

5.3 Baseline water quality impact

It is challenging to assess the environmental footprint of forestry. Nitrogen losses can be separated out across key phases in the forest cycle for farm, plantation and permanent forest. Sediment losses can be separated out across key phases in the forest cycle for farm and plantation forest, noting that in this report farm plantation forestry is forestry within a farm boundary that is planted for the purpose of harvesting. However, P losses are unable to be identified across the forestry cycle from the literature (noting a benefit of process-modelling is that if P-loss is governed or linked to erosion, then the erosional rate estimates for sediment in the FWMT can be used to guide how P loss varies over the forest cycle). In terms of contaminant footprint, it is assumed that farm and plantation forestry have the same base footprint, but they are separated out in terms of profit. There is not enough information to include the impacts of one-off weather-related events. At this stage of the FWMT build, the average impact for each contaminant across the forestry cycle is an alternative to separating environmental impacts across forestry cycles. The recommended values for N, P and sediment are summarised in Table 32, Table 33 and Table 34.

There is information available to separate out base N loss by key stages in the forestry cycle (based on studies such as Davis, 2014). It is assumed that the base N loss is the same for farm plantation forest and plantation forest. The permanent forest category has a different base N loss given the reduced demand for plant growth meaning there is more N available for leaching, as well as no additional fertiliser inputs.

Table 32: Base water quality impact estimates: N (kg N/ha/yr)

Years	Pre-plant	1-14	15-27	Post-harvest (approx. 2 yrs)	Average
Farm plantation forest	8	1	4	5	2.5
Plantation forest	8	1	4	5	2.5
Permanent forest	8	1	4 *ongoing	NA	3

There are extremely limited quantified P loss estimates from forestry in the literature, especially across the forestry cycle. The estimates presented here for the average P loss are the same for farm and plantation forest and lower for permanent forest on account of the permanent forestry cover. The question should be asked ahead of any effort to prioritise knowledge of P loss, to determine whether this will make a consequential impact on instream or to coast water quality (i.e., is forestry an important contributor to P; if not, then resolving forest cycle differences could be meaningless for the effort).

Table 33: Base water quality impact estimates: P (kg P/ha/yr)

Years	Average
Farm plantation forest	0.2
Plantation forest	0.2
Permanent forest	0.1

There is some information on sediment losses from forestry, however, these estimates (Table 34) have a high level of uncertainty. Especially that permanent forestry is the same as plantation and farm plantation forestry and that there is no difference between slopes. There is not enough data to differentiate indigenous versus plantation forestry, and therefore, it is likely the summary here is wrong in having the same result for permanent and plantation forest, based on the qualitative discussions in literature and the impact harvest has on sediment loss.

Table 34: Base water quality impact estimates: Sediment (kg/ha/yr)

Years	Mature trees (average)	Harvest (average)	Post-harvest (average)	Full rotation (average)
Farm plantation forest	235	575	1,800	875
Plantation forest	235	575	1,800	875
Permanent forest	NA	NA	NA	875

5.4 Mitigations

While there are actions described in the literature that can be undertaken within plantation forestry to reduce negative impacts on waterways, these are often not quantified for opportunity, benefit nor cost. This lack of robust quantified information makes it challenging to estimate the impact of actions whether represented as a source control or device in the FWMT, on contaminant and hydrological processes within both farm plantation forest and plantation forest types. As such, while the following mitigations are described in the literature, they cannot be robustly quantified for inclusion in the FWMT at this stage:

- Harvesting methods
- Post-harvest vegetation management
- Sediment control measures
- Fertiliser use
- Wetlands

Land use change could be included as a swap between plantation and permanent forestry, but there is not enough information to consider land use change between tree species. There is the potential to swap between the farm plantation forestry and plantation forestry to permanent forestry; however, due to the scarce literature there would be no difference in sediment yield, and the difference in N and P loss is minimal. In addition, the plantation and farm plantation forestry data include carbon income, which places a liability on the land that would disincentivise land uses other than forestry; however, a change to permanent forest could still occur.

Riparian areas could be included in the FWMT at this stage, provided Auckland Council can assess the average stream density for forestry in the region. The economic impact would be based on a proportional reduction in annual profit and the environmental impact would be based on NIWA (2010). However, given the Lakel et al. (2010) study, there is a concern that there is no difference in the sediment performance of buffers of varying widths (7.6 m to 30.6 m), so there is still significant uncertainty around this estimate. In addition, the opportunity is based on assuming the NES-PF is fully implemented and therefore there is only scope to increase buffers to greater than 5 m on streams that currently have 5 m buffers (or no buffers where not required under the NES-PF).

Given the above, source control mitigation options such as harvesting methods, post-harvest management and fertiliser use cannot be represented in the FWMT at this stage. In terms of device control mitigations, it is not appropriate to include sediment control and wetlands at this stage. Land use change (source control) could be utilised; however, as with the other rural land uses, this should be approached with caution. Riparian areas (device mitigation) are the only applicable mitigation to apply at this stage; however, further information on the opportunity for this to be applied would improve future iterations of the FWMT. All mitigations considered in this report are only applicable for farm and plantation forestry given that permanent forestry is assumed to not be subject to active management.

5.5 Next steps

There are two groups of next steps, actions which are focused on the short term and initial build of the FWMT, and actions which will require further analysis and are recommended to be built into future workplans for the FWMT. All these next steps have been discussed in more detail in preceding sections.

Initial next steps:

- Consider realigning the HRUs in the FWMT (Farm Forest, Mixed Forest, Urban Tree, Plantation Forest) with the groups for which data exists, namely permanent forestry, plantation forestry and farm plantation forestry. This would mean separating the farm forest HRU into both permanent and plantation farm HRUs.
- Use the information from Tables 31 to 34 to inform the base environmental and economic footprints for permanent forestry, plantation forestry and farm plantation forestry HRU groups within the FWMT.
- Confirm the opportunity for riparian buffer strips to be applied, including stream density per hectare of plantation forestry (and farm plantation forestry). The economic impact would be based on lost profit (opportunity cost). The environmental impact would be based on NIWA (2010) for a 5 m buffer strip as there is not enough evidence to alter this by alternative buffer widths.
- Work with experts to confirm sediment loss across forestry types, as this is a key contaminant lost from forestry activities (especially planation forestry) and currently, the best available data has the same average annual sediment loss from each forestry type.

Future work:

- Consider segregating the forestry HRUs by slope type and into key parts of the cycle for plantation (and farm plantation) forestry. This would help target mitigation opportunities to key aspects of the forestry cycle, namely the harvest period and window of vulnerability. However, the challenge in data availability would mean this would need to be supported by expert opinion. Presenting a disaggregated forestry cycle into different HRU groupings would enable more accurate classification of land use activities and therefore better estimates of base footprints, mitigation opportunities and mitigation cost and effectiveness; however, it would require significantly more evidence.
- Develop, or utilise a model such as FIF to generate, profitability metrics for the different HRUs that are specific to the Auckland region, are underpinned by discussions with foresters in Auckland and enable the assessment of mitigation cost alongside sensitivity analyses.
- Consider how best to fit two plantation forestry cycles (for radiata pine) of 27-years into a 50-year LCC. Once base economic models have been developed, a sensitivity analysis would support this assessment.
- A better understanding is needed of the current state of practices across forestry HRUs to inform selection of mitigation practices as well as the opportunity for these mitigations to be applied across forestry HRUs.
- There is limited quantified costs and benefit estimates of mitigation options for plantation forestry (including in farm plantation forestry). It is suggested that AC works with experts to assess and quantify the impacts of mitigations in forestry practices for plantation forestry.

6 References

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Appendix

NZCOP BEP Summary

This is a summarised version of the NZCOP Best Environmental Practices (NZFOA, 2015) as it relates to water quality. Some BEPs have been shortened and/or excluded. Please see the full Code for full BEPs. This list excludes the BEPs related to, operational planning, historical & heritage management, historical & heritage site discovery, fuel & oil, waste management, operational monitoring, environmental incident and forestry protection .

1. *General BEPs for forestry activities*
    a. *BEP Rules*
        i. *Follow all relevant rules and use appropriately qualified personnel.*
        ii. *Ensure important environmental values and restricted areas have been clearly identified or documented before an operation starts.*
        iii. *Communicate operational requirements verbally and in writing before starting to ensure personnel are aware of their environmental obligations*
        iv. *Avoid damage to restricted areas (e.g. protected riparian strips).*
        v. *Keep machinery out of waterways and riparian margins.*
        vi. *Wash machinery where weed transfer is an identified risk.*
        vii. *Remove all rubbish from site and dispose of in a legally acceptable manner.*
        viii. *Monitor the effects of the activity during and after the operation to ensure compliance.*
    b. *BEP Guidelines*
        i. *Undertake work in suitable weather.*
        ii. *Instigate maintenance programmes appropriate to the activity and environmental risk.*
        iii. *Consider benefits of wider riparian setbacks*

2. *Earthworks - Construction and maintenance of roads, waterway crossings, processing areas, landings, tracks, firebreaks, dams, water control structures, quarries, metal stockpiles and other engineering works.*
    a. *BEP Rules*
        i. *Design earthworks appropriate to site conditions.*
        ii. *Place debris where it will not affect sensitive areas.*
        iii. *No earthworks within 5 m of permanent waterways except at designated crossings, access points or where topographical constraints leave no alternative.*

iv. *Earthworks should be stabilised using appropriate techniques.*

v. *Do not incorporate slash or other organic material into steep fill batters.*

vi. *Install correctly designed waterway crossings, sediment traps and cut-off spacing according to site conditions.*

b. *BEP Guidelines*

i. *Programme earthworks to best fit seasonal conditions and allow stabilisation before use.*

ii. *Locate earthworks to avoid sensitive features, unnecessary disturbance and exposure of unstable areas.*

iii. *Direct water to stable areas and away from waterways (e.g. using fluming, socks, cut-offs, culverts).*

iv. *Control and filter runoff (e.g. through vegetation or slash, sediment traps, water tables and berming).*

v. *Place topsoil where it is stable and recoverable for reuse where required.*

3. *Harvesting - Includes clear-felling, road line salvage and production thinning.*

a. *BEP Rules*

i. *Do not use waterways as extraction corridors or routes.*

ii. *Install appropriate water and sediment controls and prevent runoff flowing directly into waterways (e.g. use water bars and cut-offs, sediment traps, slash redistribution, trac and soft point corduroying).*

iii. *Maintain water and sediment controls in effective operating condition until site is decommissioned.*

iv. *Use appropriate felling and extraction techniques to minimise impact in sensitive areas. Consider leaving trees standing or felling to waste if unavoidable.*

v. *Minimise tracking to reduce soil disturbance, compaction and erosion.*

vi. *Decommission the site to appropriate standard once harvesting complete.*

vii. *Have and maintain water and sediment controls post-harvest until site is revegetated, rehabilitated or otherwise stable.*

viii. *Decommission haul tracks once harvesting complete.*

ix. *Stabilise slash and bird nests after harvest.*

b. *BEP Guidelines*

i. *Use harvesting machinery that best suits the constraints of the harvest plan (e.g. use mechanical carriages capable of full log suspension over waterways).*

ii. *Use swing yarders that can operate in confined spaces reducing the need for large landings.*

iii. *Use ground based systems suited to soil type.*

iv. *Aim for extraction techniques that achieve suspension of the butt end of the log.*

v. *Avoid trimming stems in water channels, flood ways or riparian areas.*

vi. *Use debris traps where in-stream slash removal is unachievable.*

vii. *Decompact landings after use if not required for future.*

4. *Slash management - Slash and wood debris generated from harvesting operations. Comprises off-cuts from log making, and debris from felling and extraction to the landing.*

   a. *BEP Rules*

      i. *Ensure slash management requirements have been identified and clearly documented before operations start (e.g. slash storage sites).*

      ii. *Where there is insufficient space for onsite slash disposal, use temporary slash storage where slash can accumulate before being transported off site.*

      iii. *Maintain water and sediment controls in operating condition until the site is decommissioned to prevent water building up in slash piles and adjoining landings.*

      iv. *Monitor and maintain slash piles to ensure they are always stable.*

      v. *Identify alternative disposal sites if available slash storage space is exceeded.*

   b. *BEP Guidelines*

      i. *Pull back or burn landing slash where 'birds nests' are on unstable or potentially unstable ground.*

      ii. *Use directional felling or other appropriate method to minimise the amount of debris that is deposited in streams.*

      iii. *Remove as much slash and debris as practicable from intermittent streams where flood flows have potential to mobilise debris causing stream blocking, diversion or damage to downstream structures.*

      iv. *Use debris traps at strategic locations downstream where slash removal is not possible. This generally require expert consultation and resource consent.*

      v. *Ensure slash left adjacent to streams cannot be picked up by flood flows.*

5. *Waterway crossing - Includes permanent and temporary structures that cross waterways. Includes bridges, culverts, fords, battery culverts, drift decks and log crossings.*

    a. *BEP Rules*

        i. *Crossing design must consider waterway environmental values, physical attributes and intended crossing use.*

        ii. *Design crossings appropriate to site conditions and anticipated traffic use.*

        iii. *Ensure no contaminants enter waterways.*

        iv. *Avoid in-channel work during fish spawning season.*

        v. *Decommission temporary crossings after use.*

        vi. *Undertake a post-operational audit upon completion of the job.*

    b. *BEP Guidelines*

        i. *Programme earthworks to best fit seasonal conditions and stabilisation before use.*

        ii. *Confine disturbance to the immediate work site.*

        iii. *Use existing structures, where present, as a tool to assist in determining effective new structures.*

        iv. *Where possible, crossings should be located perpendicular to waterways with abutments on solid level ground on each bank.*

        v. *Earthworks should be stabilised using appropriate techniques.*

        vi. *Locate earthworks to avoid sensitive features, unnecessary disturbance and exposure of unstable areas.*

        vii. *Avoid steep approaches to and from crossings.*

        viii. *Cross where waterway banks are solid and beds stable.*

        ix. *Use slash racks to protect culverts where slash build up is expected*

        x. *Divert road/track runoff away from crossings using berms, cutouts, culverts or flumes.*

        xi. *Use bridges or low level crossings on larger waterways.*

        xii. *Construct fords for infrequent vehicle use, and to cross waterways that have hard streambeds, low flows and low in-stream values.*

6. *Mechanical land preparation - The preparation of land for planting using machinery including spot mounding/cultivation, line-raking, wind rowing, root and slash raking, v-blading, ripping and mounding, roller crushing, mulching and conversion from plantations.*

    a. *BEP Rules*

i. Minimise soil disturbance except where v-blading and ripping and mounding are being carried out to ameliorate specific adverse soil properties.

ii. Have and maintain water and sediment controls until site is revegetated, rehabilitated or otherwise stable.

iii. Ensure sediment runoff is contained within the work site and does not directly runoff into waterways.

iv. Undertake a post-operational audit on completion of the job.

b. BEP Guidelines

i. Leave an undisturbed buffer strip of at least 5 m adjacent to permanently flowing streams.

ii. Cultivate or rip landing sites across the slope and ensure water does not accumulate in fill areas.

iii. Operate along the contour to minimise runoff being concentrated down cultivated lines. Where unavoidable, limit downhill runs to a maximum continuous length of 50 m.

iv. Where soil properties and rainfall predispose, align slash windows along the contour of sloping land and within broad valley floors to help trap and filter sediment.

v. Blade or rake at least one line on the contour along the lower boundary of operations to help contain sediment within the work site and prevent runoff concentration at low points or gullies.

7. Agrichemical application

a. BEP Rules

i. Ensure ground-based equipment and/or aircraft loading and mixing areas are away from any streams or water supplies and the site is suitably located/bunded to contain spills to the immediate area.

ii. Disperse any residual chemical mixture over the target area at or below standard concentrations rather than dumping in a concentrated quantity over a small area.

iii. Undertake a post-operational audit on completion of the job.

b. BEP Guidelines

i. Use GPS technology and ensure GPS coordinates are provided and/or fly the boundary with the operator's supervisor as part of the pre-operational briefing.

ii. Utilise smoke bombs or similar to test for inversion and inappropriate wind conditions.

iii. Apply agrichemicals manually where there is a risk of spray drift.

8. *Burning*

   a. *BEP Rules*

      i. *Develop a burn prescription containing explicit requirements that are clearly mapped and documented (e.g. waterbodies, buffer zones).*

      ii. *Ensure manufacturers label recommendations are followed when using accelerants.*

      iii. *Ground-based equipment and/or aircraft loading and mixing areas are well away from streams or water supplies and are suitably located or sufficiently bunded.*

   b. *BEP Guidelines*

      i. *Consider post burn over-sowing of areas that are prone to erosion.*

      ii. *Consider burning only if alternative methods are not practical or cost-effective.*

      iii. *Undertake a post-operational audit on completion of the job.*

9. *Planting*

   a. *BEP Rules*

      i. *Meet setback requirements around restricted areas (e.g. protected riparian strips).*

      ii. *Leave a horizontal setback of at least 5m each side of all permanently flowing streams.*

      iii. *Do not plant where harvesting will not be possible without serious adverse effects.*

      iv. *Undertake a post-operational audit upon completion of job.*

   b. *BEP Guidelines*

      i. *Consider conducting a social and environmental impact assessment when planting new areas outside existing forest boundaries.*

      ii. *Increase riparian setbacks where topographical, reserve features, stream size or sensitive boundaries and identifiable future harvesting complications indicate greater margins are needed.*

10. *Pruning and waste thinning*

   a. *BEP Rules*

      i. *Do not leave slash where it could divert or block a permanent waterway, water tabl or water control.*

      ii. *Undertake a post-operational audit upon completion of job.*

   b. *BEP Guidelines*

i. Place thinning and pruning slash behind the first row of trees within the stand boundary.

ii. Fell trees away from sensitive areas.

iii. Progressively remove slash and debris rather than leaving it to when the operation is finished.

11. Fertiliser application

a. BEP Rules

i. Store fertiliser in suitably located sites (e.g. well away from waterways).

ii. Undertake a post-operational audit upon completion of job.

b. BEP Guidelines

i. Consider use of granulated slow-release fertiliser where available and cost-effective.

ii. Use aircraft with GPS navigation and control systems to ensure accurate application.

iii. Consider the use of helicopters with controlled spread hoppers in preference to fixed wing craft and gravity feed spreaders to achieve greater accuracy of application and evenness of spread.

iv. Leave buffer zones around water bodies.

v. Apply fertiliser utilising a wind direction blowing away from buffer zones and waterways.

vi. Apply fertiliser manually where there is a risk of fertiliser drift into waterbodies.