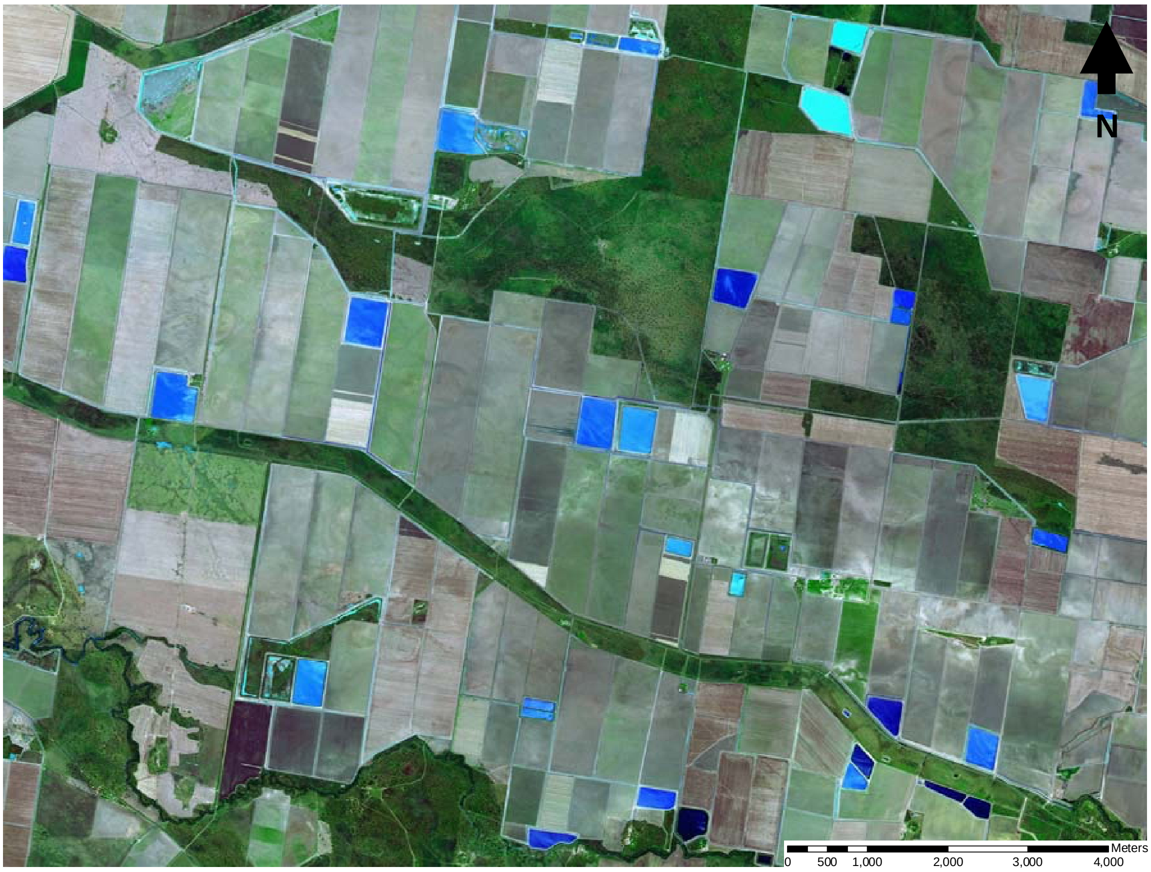
Biodiversity and Ecosystem Services Associated with Remnant Native Vegetation in an Agricultural Floodplain Landscape

Rhiannon Smith
B Nat Res (Hons) University of New England



A thesis submitted for the degree of Doctor of Philosophy of the
University of New England

June 2010

Certification

I certify that the substance of this thesis has not already been submitted for any degree and is not currently being submitted for any other degree or qualification.

I certify that any help received in preparing this thesis and all sources used have been acknowledged in this thesis.

R Smith

Rhiannon Smith

Abstract

Biodiversity, ecosystem service provision and human well-being are inextricably linked. The current rate of biodiversity loss worldwide is impacting on ecosystem service provision with negative implications for human well-being. Little quantitative information is available about the provision of most ecosystem services by most ecosystems, the effect of management on the ability of vegetation to provide services, or trade-offs in service provision with land use. This information is particularly important in agricultural landscapes where the extent of landscape change is affecting biodiversity and ecosystem service provision substantially and thus agricultural sustainability.

This study quantified the provision of carbon storage, erosion mitigation and biodiversity conservation services by five vegetation communities (river red gum *Eucalyptus camaldulensis* riparian forests, coolibah *E. coolabah* woodlands and open-woodlands, myall *Acacia pendula* tall shrublands and tall open-shrublands, black box *E. largiflorens* woodland and open-woodland, and mixed grassland – low open-chenopod shrubland) common on the lower Namoi floodplain in northern New South Wales, Australia. Sites represented the full range of structural and compositional variants encountered within each vegetation type over the 7100 km^2 study region, from heavily grazed derived grasslands to old-growth woodland or forest evidently little affected by anthropogenic disturbance.

The environmental conditions dictating the location of each vegetation type in the landscape were investigated. The distribution of vegetation types depended predominantly on soil type, flood patterns and the interaction between the two. Woody and non-woody vegetation was mapped across the study region using unsupervised classification of ten single-date SPOT 5 scenes with 85% accuracy. Woody vegetation covered approximately 7% of the lower Namoi floodplain.

Carbon storage was measured or estimated for soils, woody vegetation, dead standing vegetation, coarse woody debris, herbaceous vegetation, litter and roots. River red gum sites were the most valuable vegetation type for carbon storage, having up to 4.5% carbon content in the surface 0–5 cm soil depth increment, with total site carbon storage averaging 216 t C ha^{-1}. The most carbon-dense site was east of Narrabri and dominated by river red gum. Grasslands were the least carbon-dense with 40.0 t C ha^{-1}. The greatest proportion of carbon in river red gum sites was in woody biomass, but in all other vegetation types and especially grasslands, the top 0–30 cm of the soil was the most C-rich component of the ecosystem. Woody biomass C was positively correlated with C derived from dead standing wood, coarse woody debris and litter, but not herbaceous biomass C, which was negatively correlated.

Herbaceous vegetation cover, litter cover and macroaggregate stability as determined by the topsoil C:N ratio was used to rank sites for erosion mitigation service provision. Erosion mitigation value was assessed in terms of aggregate stability, which was determined by a relationship between mean weight diameter of aggregates and soil C:N ratio, as well as dominant cations on the clays. Soils with higher C:N contained more stable macroaggregates, and tended to be dominated by river red gum. High aggregate stability in river red gum sites was attributed to large inputs of eucalypt litter and coarse

woody debris. Highest microaggregate stability was also observed in river red gum sites and attributed to the dominance of Ca^{2+} rather than Na^{+} on clay exchange sites.

Vascular plant and bird conservation value of sites was determined by ranking sites according to the number of rare (i.e. infrequently observed) species present. For birds, species richness was also taken into account. River red gum sites were ranked highest for vascular plant and bird conservation value because they contained the highest abundance of rare species of both vascular plants and birds. However, river red gum sites also contained the greatest number of introduced plant species presumably as a result of flood mediated dispersal of propagules. All vegetation communities were included among the sites of highest conservation value for both vascular plants and birds. However, in the top 30% (16 of 54) of sites ranked according to conservation value, only five sites were valuable for both plant and bird conservation. River red gum sites had the most structurally complex vegetation, which coupled with their proximity to water, encouraged high bird species richness and abundance. Woody plants were the most influential vegetation component determining bird conservation value, but different vegetation types were preferentially used by different bird species, implying that the full spectrum of vegetation types is required to maximise bird and plant conservation at the regional scale.

Increasing grazing intensity severely diminished both plant and bird conservation value at river red gum and coolibah sites as a result of the loss of rare species. Grazing also detracted from carbon storage, both directly through biomass consumption and indirectly through associated management (such as ring-barking to increase herbaceous

biomass production and clearing). The functional richness (i.e. the number of different life-forms of vegetation types) was more influential than species richness in terms of ecosystem service provision. Shannon–Wiener diversity of vegetation communities showed no relationship with ecosystem service provision. No trade-offs were evident between the three ecosystem services measured in this study, but conservation value and carbon services declined under increasingly intense grazing. Increasing woody vegetation biomass and cover resulted in decreased herbaceous biomass production, leading to the trade-off between nature conservation and carbon sequestration on the one hand, and livestock production on the other. There are few ungrazed sites in the study region, hence natural capital may be diminished still further with continued grazing of almost the entire landscape.

Acknowledgements

Firstly, I am indebted to my supervisors, Nick Reid, David Tongway, Paul Frazier and Guy Roth for their friendship, patience and encouragement, and for their generous contribution of time and knowledge.

Many landholders allowed access to their properties, provided insightful knowledge of the region and helped with equipment housing and repairs during fieldtrips. In particular, the Swansbra family and everyone at 'Lammermoor' provided a home away from home during many weeks of fieldwork, and really went above and beyond to help out in every way possible. Their generous hospitality, support and enthusiasm towards the project were gratefully appreciated. John and Robyn Watson at 'Kilmarnock' also provided accommodation and looked after us during field trips, and were involved in shaping the direction of the project in the very early days.

Numerous people bravely helped out with fieldwork (some on multiple trips). I am extremely thankful for their assistance and hope the experience hasn't scared them too much. In particular, Nick Reid, Rohan Smith, Stuart Green, Meg Good, Scott Purcell, Peter Berney, Fiona Norrie, Tegan Smith, Mark Dahm, Guy Roth, Evan Cleland, Stacey Spanswick, Hamish Caddy, Sian Smith and Christopher Hardie helped out immensely.

The support staff at UNE: Paul Lisle, Stuart Green, Cate MacGregor, Dan Alter, Leanne Lisle, Marion Costigan, Jim O'Neill, Bec Wood, Jo Porter, Chris Cooper and Shirley Fraser kept things ticking, and John Hunter, Dorothy Bell and Wal Whalley assisted with plant identification. David Tongway, Megan Purvis and Rohan Smith helped in the

lab. Brian Wilson, Dave Carr, Jackie Reid and Rick Young offered helpful advice along the way, as did Alan Ede, Warick Browne, Ian Oliver, Chris Nadolny, Des Lang, Sean Murphy, Tom Lewis, Michael Drielsma, Simon Ferrier, Rob Loch and Will Dorrington.

Thanks to the postgrads, postdocs and academics in EcoMan who made my time here enjoyable and memorable, especially those that I shared an office with. Your humour, support and sympathy helped me through some very long days. Matt Tighe and Carlos Muñoz helped me out on so many levels on so many occasions. Matt's knowledge of all things soil related is amazing and his ability to make things happen in R is almost magical, while Carlos has a real gift with ArcGIS and both have been good for a cup of coffee and at time, a whinge. Special mention must be made of Subhashni Taylor, Melissa Danks, Morag Stewart, Nick Schultz and Megan Good; I am so grateful for their help with sorting out referencing nightmares in the final few weeks.

Finally, my family and friends supported me in every sense of the word and I could not have done this without them. Though they were all so far away they regularly made it known that they were around, and encouraged me the whole time.

Financial support for this project was provided by the Cotton Catchment Communities CRC, the Cotton Research and Development Corporation and the Namoi Catchment Management Authority. Substantial in-kind support was also provided by the Department of Environment, Climate Change and Water.

Sorry if I have forgotten anyone!

Note to Examiners

This thesis has been written in chapter format, but in a form that is amenable to conversion to papers for publication at a later date. While I have tried to reduce the duplication of material among chapters to a minimum, some repetition between chapters remains. Please excuse this repetition.

Table of Contents

Certification i
Abstract ii
Acknowledgements iv
Note to Examiners viii
Contents ix
List of Figures xvi
List of Tables xx

Chapter 1 – Introduction 1
1.1 Introduction 2
1.2 Ecosystem Services: The Concept 5
1.3 Why is the Ecosystem Services Concept Topical? 6
1.4 Classification of Ecosystem Services 8
1.5 Vegetation as an Ecosystem Service Provider 11
1.6 Vegetation Condition 14
1.7 The Australian Cotton Industry 18
1.8 Ecosystem Services and Cotton Production 21
1.9 Research Gaps and Thesis Aims 23
1.10 Thesis Outline 25

Chapter 2 – What Drives Landscape Scale Vegetation Patterns on the Fertile Semi-arid Floodplains of Northern NSW? 27
2.1 Introduction 28
    2.1.1 Aims and Objectives 31
2.2 Study Region 31
    2.2.1 Location 31
    2.2.2 Climate 33
    2.2.3 Landform, Geology, Topography and Drainage 35
    2.2.4 Soils 39
    2.2.5 Human Occupation, Land Use and Vegetation Management 41
2.3 Current Study 45
    2.3.1 Field and Laboratory Methods 45
    2.3.2 Statistical Analysis 48
2.4 Results 50
2.5 Discussion 57
    2.5.1 Vegetation of the Lower Namoi Floodplain 57
        2.5.1.1 River Red Gum 57
        2.5.1.2 Coolibah 59
        2.5.1.3 Myall 61
        2.5.1.4 Black Box 63

2.5.1.5 Grasslands 63
2.5.2 Further Research 66
2.6 Conclusions 68

Chapter 3 – Single-date SPOT 5 Imagery Accurately Maps Woody Vegetation Extent on a Semi-arid Floodplain 69

3.1 Introduction 70
3.2 Methods 73
3.2.1 Study Region 73
3.2.2 Satellite Data 75
3.2.3 Study Sites 77
3.2.3.1 Site 1 (Narrabri) 77
3.2.3.2 Site 2 (Pilliga) 78
3.2.3.3 Site 3 (Curlewis) 79
3.2.4 Image Analysis 80
3.2.4.1 Classification of Woody Vegetation Communities 80
3.2.4.2 Selecting the Optimal Number of Unsupervised Classes 81
3.2.4.3 Catchment-scale Tree Cover Mapping 82
3.3 Results 83
3.3.1 Unsupervised Classification of Tree Species 83
3.3.2 Optimal Unsupervised Class Numbers 84
3.3.3 Catchment-scale Tree Cover Mapping 85
3.4 Discussion 87
3.4.1 Timing of Data Capture 87
3.4.2 Spectral, Physical and Structural Similarity of Vegetation Types 88
3.4.3 Time and Increasing Class Number 89
3.4.4 Implications for Environmental Management 90
3.5 Conclusions and Future Research 90

Chapter 4 – Carbon Storage Value of Native Vegetation on the Lower Namoi Floodplain, NSW 92

4.1 Introduction 93
4.1.1 Carbon Storage and an Ecosystem Service 94
4.1.2 Vegetation Condition for Carbon Storage 96
4.1.2.1 Above-ground Woody Vegetation 97
4.1.2.2 Above-ground Herbaceous Vegetation 98
4.1.2.3 Litter 100
4.1.2.4 Coarse Woody Debris (CWD) and Dead Standing Wood 101
4.1.2.5 Soil and Plant Roots 102

4.1.3 Research Gaps 104
4.1.4 Aims and Objectives 105
4.2 Methods 105
4.2.1 Study Region 105
4.2.2 Field Sampling 107
4.2.2.1 Above-ground Woody Vegetation Biomass 107
4.2.2.2 Litter and Herbaceous Above-ground Vegetation Biomass 108
4.2.2.3 CWD 109
4.2.2.4 Soils 110
4.2.2.5 Roots and Total Site C 111
4.2.3 Statistical Analysis 111
4.3 Results 112
4.3.1 Above-ground Woody Vegetation 112
4.3.2 Above-ground Herbaceous Vegetation 113
4.3.3 Litter 114
4.3.4 Dead Standing Wood 115
4.3.5 CWD 116
4.3.6 Above-ground Biomass 117
4.3.7 Roots 118
4.3.8 Soil 119
4.3.9 Total Site Carbon 121
4.3.10 Vegetation Influences 123
4.4 Discussion 123
4.4.1 Above-ground Woody Vegetation 124
4.4.2 Limitations of Allometric Equations in Estimating Tree Biomass 126
4.4.3 Above-ground Herbaceous Vegetation 127
4.4.4 Litter 128
4.4.5 CWD and Standing Dead Wood 129
4.4.6 Soil and Roots 132
4.4.7 Influence of Vegetation Condition on Total Site Carbon 133
4.5 Conclusion and Future Research 134

Chapter 5 – Erosion Mitigation Value of Native Vegetation of the Lower Namoi Floodplain, NSW **136**
5.1 Introduction 137
5.2 Methods 141
5.2.1 Study Region 143
5.2.2 Field Sampling 145
5.2.3 Soil Analyses 145
5.2.4 Statistical Analyses 147
5.2.4.1 Soil and Vegetation Type Characteristics 147

5.2.4.2 Water Stable Aggregation 148
5.2.4.3 ASWAT 149
5.3 Results 150
5.3.1 Surface and Subsurface Soil Characteristics 150
5.3.2 Characteristics of Vegetation Types 151
5.3.3 Water Stable Aggregation 152
5.3.4 Vegetation Influence on Water Stable Aggregation 154
5.3.5 ASWAT – Site Level 155
5.3.6 ASWAT – Cover Class Interactions 161
5.4 Discussion 162
5.4.1 Water Stable Aggregation 163
5.4.2 Vegetation Types and Water Stable Aggregation 166
5.4.3 ASWAT 167
5.4.4 Cover Classes 170
5.4.5 Influence of Vegetation and Different Vegetation Types on Soil Aggregate Stability 172
5.4.6 Management Implications 173
5.5 Conclusion 174

Chapter 6 – Biodiversity Conservation Value: Variation with Vegetation Type and Grazing Management on the Lower Namoi Floodplain, NSW **175**
6.1 Introduction 176
6.1.1 The Value of Native Vegetation in Agricultural Landscapes 177
6.1.2 Biodiversity in Agricultural Landscapes 177
6.1.3 Grazing Impacts on Plant and Bird Communities 181
6.1.4 Conservation Assessment 183
6.1.5 Indicators of Biodiversity Conservation Value 185
6.1.6 Aims and Objectives 186
6.2 Methods 188
6.2.1 Study Region 188
6.2.2 Vegetation 190
6.2.2.1 River Red Gum 190
6.2.2.2 Coolibah 191
6.2.2.3 Myall 191
6.2.2.4 Black Box 192
6.2.2.5 Native and Derived Grasslands 192
6.2.3 Field Sampling 193
6.2.3.1 Environmental Characteristics 193
6.2.3.2 Vegetation 194
6.2.3.3 Birds 194
6.2.3.4 Habitat Characteristics 195
6.2.3.5 Grazing Intensity 196

6.2.4 Data Analysis 196
6.3 Results 200
6.3.1 Vegetation 200
6.3.2 Birds 211
6.4 Discussion 221
6.4.1 Efficiency of Different Measures of Biodiversity Conservation Value 221
6.4.2 Vegetation Type and Environmental Influences on Plant Conservation Value 222
6.4.3 Grazing Impacts on Site Value for Plant Conservation 227
6.4.4 Vegetation Type and Habitat Influences on Bird Conservation Value 231
6.4.5 Grazing Impacts on Site Value for Bird Conservation 235
6.4.6 Overall Conservation Value of Sites 237
6.5 Conclusion 237

Chapter 7 – Comparison of Ecosystem Services provided by Different Vegetation Types and Condition States: Trade-offs with Management and Land Use Intensity **239**
7.1 Introduction 240
7.1.1 Biodiversity and Ecosystem Service Provision 242
7.1.2 The 'Ecosystem Service Providers' Concept 243
7.1.3 Vegetation Condition Assessment for Ecosystem Service Provision 244
7.1.4 Grazing Effects on Ecosystem Service Provision 245
7.1.5 Ecosystem Service Provision on the Inland Floodplains of the Namoi River 247
7.1.6 Aims and Objectives 247
7.2 Methods 248
7.2.1 Study Region 248
7.2.2 Ecosystem Service Data 250
7.2.3 Grazing Intensity 251
7.2.4 Statistical Analyses 251
7.3 Results 252
7.4 Discussion 258
7.4.1 Relationship between Biodiversity and Ecosystem Service Provision 258
7.4.2 Influence of Vegetation Functional Richness on Ecosystem Service Provision 260
7.4.3 Ecosystem Service Providers (ESPs) for Bird Conservation, Carbon Storage and Erosion Mitigation Services 262
7.4.4 Vegetation Condition Assessment for Ecosystem Service Provision 264

7.4.5 Influence of Grazing Intensity on Ecosystem Service Provision 267
7.4.6 Trade-offs between Grazing and Ecosystem Service Provision 269
7.5 Conclusion 271

Chapter 8 – Synthesis and Conclusions **272**
8.1 Introduction 273
8.1.1 Aims and Objectives 275
8.2 Summary of Main Findings 276
8.2.1 Vegetation Communities of the Lower Namoi Floodplain 276
8.2.2 Woody Vegetation Mapping 278
8.2.3 Carbon Storage Value of Lower Namoi Floodplain Vegetation 279
8.2.4 Erosion Mitigation Value of Lower Namoi Floodplain Vegetation 281
8.2.5 Biodiversity Conservation Value of Lower Namoi Floodplain Vegetation 282
8.2.6 Congruence between Plant Species Richness and Functional Richness and Ecosystem Service Provision 283
8.2.7 Ecosystem Service Providers Responsible for Generation of Ecosystem Services 284
8.2.8 Impact of Grazing on Ecosystem Service Provision 284
8.3 Synthesis 285
8.4 How the Research Adds to Knowledge 289
8.5 How Research can contribute to Resource Management Practice 290
8.6 Management Recommendations 292
8.7 Future Research 294
8.7.1 Vegetation Community Distribution 294
8.7.2 Vegetation Mapping 294
8.7.3 Carbon Storage 295
8.7.4 Erosion Mitigation 296
8.7.5 Biodiversity Conservation 297
8.7.6 Functional Richness Impacts on Ecosystem Service Provision 297
8.7.7 Other Ecosystem Services 298

References **299**
Appendix 1: Native, long-lived perennial plant species recorded during surveys of the lower Namoi floodplain. 344
Appendix 2: Pearson correlation matrix for MWD of aggregates and all other physico-chemical variables for nine surface (0–5 cm) soil samples. 348
Appendix 3: Pearson correlation matrix for MWD of aggregates and all other physico-chemical variables for 16 subsurface (20–30 cm) soil samples. 349

Appendix 4: Spearman correlations between ASWAT and all physico-chemical variables describing surface soil (0–5 cm) samples from 60 sites. 350
Appendix 5: Spearman correlations between ASWAT and all physico-chemical variables describing the subsurface soils (20–30 cm) at 60 sites. 351
Appendix 6: Spearman rank correlations between native plant species richness ranking, maximum rarity ranking, NMDS axes, the five principal components and individual environmental variables. 352
Appendix 7: Spearman rank correlations between native bird species richness ranking, maximum rarity ranking, NMDS axes, the five principal components and individual habitat variables. 353
Appendix 8: Average abundance and frequency of occurrence (%) of vascular plant species in five vegetation types common to the lower Namoi floodplain 354
Appendix 9: Average abundance and frequency of occurrence (%) of bird species in five vegetation types common to the lower Namoi floodplain 363

List of Figures

Figure 1.1	Millennium Ecosystem Assessment diagram depicting the strength of linkages between categories of ecosystem services and components of human well-being	8
Figure 1.2	Distribution of cotton production in the lower Namoi catchment. Note the concentration of cotton production in riparian and floodplain areas between Narrabri and Burren Junction (data from DLWC 1996).	19
Figure 2.1	Namoi catchment in northern NSW showing the extent of the floodplain.	32
Figure 2.2	The Darling Riverine Plains, Brigalow Belt South, Nandewar and New England Tablelands bioregions dissect the Namoi catchment.	33
Figure 2.3	Mean annual rainfall across the Namoi catchment.	34
Figure 2.4	Long-term mean monthly rainfall and first and ninth decile data collected at Walgett since 1878, Narrabri 1891 and Gunnedah 1877 (Bureau of Meteorology 2008).	34
Figure 2.5	Mean daily maximum and minimum temperatures at Walgett (data collected 1878–1993), Narrabri (1962–2002) and Gunnedah (1876–2006) (Bureau of Meteorology 2008).	35
Figure 2.6	Digital Elevation Model of the Namoi catchment showing selected major towns and rivers.	36
Figure 2.7	Coolibah (*Eucalyptus coolabah*) trees at Walgett ringbarked during the early 1900s.	44
Figure 2.8	Lower Namoi floodplain study region in northern NSW showing location of study sites.	46
Figure 2.9	DCA ordination illustrating differences in dominant life-forms of native, long-lived, perennial vegetation for five vegetation types on the lower Namoi floodplain.	54
Figure 2.10	DCA ordination of the relationship between environmental variables and different plant life forms in the native, long-lived, perennial component of the vegetation on the lower Namoi floodplain.	55
Figure 2.11	CCA ordination showing the environmental variables driving long-lived, native, perennial vegetation patterns across the lower Namoi floodplain.	56
Figure 2.12	DCA biplot showing distribution of common grass species in relation to soil and rainfall gradients.	65
Figure 3.1	Namoi Catchment in northern NSW indicating extent of floodplain study region (defined by largest recorded flood) and study sites.	74
Figure 3.2	SPOT 5 image of study site 1 located 10 km east of Narrabri, NSW, indicating major landmarks, land uses and vegetation types (image date 1 January 2005).	77

Figure 3.3 a) *Eucalyptus camaldulensis* b) *Eucalyptus populnea* c) *Eucalyptus melliodora* dominated woodlands at site 1. 78

Figure 3.4 SPOT 5 image of study site 2 located 20 km north-west of Pilliga, NSW, indicating major landmarks, land uses and vegetation types (image date 25 November 2004). 79

Figure 3.5 SPOT 5 image of study site 3 west of Curlewis, NSW, indicating major landmarks, land uses and vegetation types (image date 26 May 2005). 80

Figure 3.6 Accuracy achieved (mean and range) with increasing number of classes for woody and non-woody landcover categories. 84

Figure 3.7 Woody vegetation of the Namoi Floodplain as mapped using unsupervised classification of SPOT 5 data. 86

Figure 4.1 Major C fluxes in an ecosystem. The net accumulation of C in an ecosystem is equal to GPP less losses due to plant and heterotrophic respiration, disturbance, and leaching (adapted from Chapin et al. 2002). 95

Figure 4.2 Namoi floodplain in northern NSW, Australia. The study region was the lower Namoi floodplain from Boggabri downstream to Walgett. 106

Figure 4.3 Carbon stored in above-ground woody vegetation for six vegetation types ($F = 33.1$, $df = 5$, $P < 0.001$). Letters represent significant differences (LSD, $P < 0.05$). 113

Figure 4.4 Carbon stored in above-ground herbaceous vegetation for six vegetation types ($F = 2.54$, $df = 5$, $P = 0.039$). Letters represent significant differences (LSD, $P < 0.05$). 114

Figure 4.5 The C stored in litter in six vegetation types ($F = 7.65$, $df = 5$, $P < 0.001$). Letters represent significant differences (LSD, $P < 0.05$). 115

Figure 4.6 Dead standing C in six vegetation types ($F = 6.05$, $df = 5$, $P < 0.001$). Letters represent significant differences (LSD, $P < 0.05$). 116

Figure 4.7 The C content of CWD in six vegetation types ($F = 10.8$, $df = 5$, $P < 0.001$). Letters represent significant differences (LSD, $P < 0.05$). 117

Figure 4.8 Total above-ground C for six vegetation types ($F = 32.2$, $df = 5$, $P < 0.001$). Letters represent significant differences (LSD, $P < 0.05$). 118

Figure 4.9 Root C for six vegetation types ($F = 27.9$, $df = 5$, $P < 0.001$). Letters represent significant differences (LSD, $P < 0.05$). 119

Figure 4.10 Soil organic C in the top 30 cm for six vegetation types ($F = 8.04$, $df = 5$, $P < 0.001$). Letters represent significant differences (LSD, $P < 0.05$). 120

Figure 4.11 Mean TOC distribution by vegetation type ± 1 S.E. 121

Figure 4.12 Mean total site C by vegetation type ($F = 26.7$, $df = 5$, $p < 0.001$). Letters indicate significant differences (LSD, $P < 0.05$). 122

Figure 4.13 Proportion of mean total site C contributed by each ecosystem 122

component.
Figure 4.14 (a) A river red gum site largely unaffected by dieback and containing the greatest woody vegetation biomass of the sites surveyed. (b) An intermediate site in the dieback – woody biomass continuum: the trees at this site are severely stressed but most are still alive. (c) A site severely affected by dieback: many of the trees in this site are dead and the least woody biomass of river red gum sites was found at this site. 125
Figure 4.15 The branching habit of myall trees leads to production of large CWD as the canopy becomes too heavy, causing the trunk to split at the lowest fork. 131
Figure 5.1 Location of the Namoi floodplain study region in northern NSW. 142
Figure 5.2 Linear regression relationships between MWD in surface soil samples ($n = 9$) and (a) TOC ($MWD = 126.49*TOC – 21.79$ ($R^2 = 0.5311$, $P = 0.016$) and (b) C:N ($MWD = 109.08*C:N – 813.41$ ($R^2 = 0.7626$, $P = 0.001$). 153
Figure 5.3 PCA of 18 surface soil variables (normalised and standardised) for 60 sites plotted against PC1 and PC2. Bubbles represent median ($n = 9$) ASWAT score for each site, larger bubbles indicate more dispersive soils. 156
Figure 5.4 Two dimensional PCA ordination of 19 surface soil variables (normalised and standardised) for 60 sites representing six vegetation types commonly found on vertosols on the lower Namoi floodplain. 157
Figure 5.5 Subsurface ASWAT score ($n = 60$) plotted against log (Na(cmol(+)/kg)+1) showing the fitted model ($ASWAT \sim 12.53*\log(Na+1) + 0.375$) ($R^2 = 0.4953$, $P < 0.001$) and 95% confidence intervals. 158
Figure 5.6 PCA ordination of 18 subsurface soil variables (normalised and standardised) and 60 sites against PC1 and PC2. Bubbles represent the median ($n = 9$) ASWAT score at each site, larger bubbles represent more dispersive soils. 159
Figure 5.7 Two dimensional PCA ordination of 19 subsurface soil variables (normalised and standardised) for 60 sites representing six vegetation types commonly found on vertosols on the lower Namoi floodplain. 160
Figure 5.8 Median surface soil ASWAT score for six cover classes ($F_{5,543} = 6.97$, $P < 0.001$). Letters indicate significant differences (Tukey's HSD $P < 0.05$). 161
Figure 5.9 Proportion of cores representing each cover class in each vegetation type. 162
Figure 6.1 Namoi catchment in northern NSW showing IBRA boundaries and the Namoi floodplain. 188
Figure 6.2 Namoi catchment land use showing the extent of agricultural development and location of important conservation reserves. 189
Figure 6.3 Distribution of sample frequency for 181 plant species. 201
Figure 6.4 PCA ordination of environmental variables by vegetation type. 203

Figure 6.5 NMDS ordination of plant community composition at 54 sites by (a) vegetation type (derived grasslands are coloured black) and grazing intensity in (b) river red gum and (c) coolibah sites. 205

Figure 6.6 NMDS ordination of 54 sites based on plant composition, with a) native plant species richness and b) maximum rarity inverse rankings superimposed (larger bubbles represent higher native species richness and maximum rarity). 206

Figure 6.7 CAP analysis showing the relationship between vegetation type and vascular plant composition and a gradient of available moisture represented by PC1 ($\delta_1^2 = 0.86$, $p < 0.001$). Derived grasslands are shown in black. 207

Figure 6.8 Site ranking for native species richness and maximum rarity (n = 54) by (a) vegetation type and (b) grazing intensity. 208

Figure 6.9 NMDS bubble plot showing number of (a) grazing-sensitive and (b) grazing-tolerant species at 54 sites. 210

Figure 6.10 Distribution of sample frequency for 87 bird species. 212

Figure 6.11 PCA ordination of habitat characteristics by vegetation type. 214

Figure 6.12 NMDS ordinations of (a) bird community composition at 54 sites by vegetation type, (b) 12 river red gum sites by grazing intensity and (c) 16 coolibah sites by grazing intensity. 216

Figure 6.13 NMDS ordination of bird assemblages at 54 sites showing a) bird species richness and b) site maximum rarity. 217

Figure 6.14 CAP analysis of bird composition showing the relationship between vegetation type and PC1 which represents a gradient in canopy cover and related variables ($\delta_1^2 = 0.76$, $p = 0.001$). 218

Figure 6.15 Site ranking for native species richness and maximum rarity (n = 54) by (a) vegetation type and (b) grazing intensity. 219

Figure 6.16 NMDS bubble plot showing number of (a) grazing-sensitive and (b) grazing-tolerant species present at different sites. 221

Figure 7.1 Namoi catchment in northern NSW showing the extent of the floodplain. The floodplain between Boggabri in the east and Walgett in the west forms the study region for this study. 249

Figure 7.2 (a) PCA of ESPs with (b) bird conservation value, (c) carbon storage value and (d) erosion mitigation value represented by increasing bubble size. 256

Figure 7.3 Synergies and trade-offs between ecosystem services. 270

Figure 8.1 Conceptual diagram of the relationships between abiotic and societal influences on ESPs and provision of three ecosystem services. 286

Figure 8.2 Changes in vegetation cover and biomass in a grassland site north-east of Pilliga during drought (spring 2007) and after above-average rainfall (spring 2008). 287

List of Tables

Table 1.1 Ecosystem services provided by terrestrial ecosystems (Reid 2010). 13

Table 2.1 Mean values for soil variables for each vegetation type. 51

Table 2.2 Results of the SIMPER analysis of long-lived, perennial, native vegetation. Species are indicative of each vegetation type. 52

Table 2.3 Results from DCA of the native, long-lived perennial flora, and environmental (rainfall and soil) variables at 57 sites across the lower Namoi floodplain. 53

Table 2.4 Results from CCA for native, long-lived, perennial flora, and environmental (rainfall and soil) variables at 57 sites across the lower Namoi floodplain. 55

Table 3.1 Dominant woody vegetation types of the mid–lower Namoi floodplain. 75

Table 3.2 Satellite images used to map woody vegetation extent across the mid–lower Namoi floodplain. 76

Table 3.3 Error matrix and accuracy assessment for unsupervised classification at site 1 using 200 classes. 83

Table 3.4 Error matrix and accuracy assessment for woody vs non-woody vegetation Namoi floodplain. 85

Table 4.1 Site mean and maximum DBH of the dominant species by vegetation type 112

Table 4.2 Spearman correlation matrix for C contribution in different ecosystem components (n = 61). 123

Table 5.1 Mean surface soil variables (0–5 cm) for each vegetation type. 150

Table 5.2 Mean subsurface soil variables (20–30 cm) for each vegetation type. 151

Table 5.3 Mean litter, herbaceous vegetation and canopy cover (%), and litter and herbaceous vegetation biomass (kg DM/ha) for each vegetation type. 152

Table 5.4 Spearman correlations between selected vegetation variables and surface and subsurface soil characteristics influencing water stable aggregation (n = 60). 155

Table 6.1 Mean plant species richness and conservation value rankings for each vegetation type. 201

Table 6.2 Mean plant species richness and conservation value rankings for each grazing intensity. 202

Table 6.3 Efficiency of five criteria for ranking sites according to vascular plant conservation value (16 highest ranked sites). 203

Table 6.4 Five principal components representing a linear combination of 15 environmental variables and the variance explained by each. 204

Table 6.5 Spearman rank correlations between native species richness ranking, maximum rarity ranking, NMDS axes and five principal components. 207

Table 6.6 Grazing-sensitive species (SIMPER), their contribution to Bray–Curtis dissimilarity and abundance at regularly (n = 13) and rarely (n = 9) grazed river red gum and coolibah sites. 209

Table 6.7 Grazing-tolerant species identified by SIMPER with their contribution to Bray–Curtis dissimilarity and mean abundance at regularly (n = 13) and rarely (n = 9) grazed river red gum and coolibah sites. 210
Table 6.8 Mean bird species richness and conservation value rankings for each vegetation type. 211
Table 6.9 Efficiency of five criteria for ranking sites according to bird conservation value (16 highest ranked sites). 212
Table 6.10 Mean bird species richness and conservation value rankings for each grazing intensity. 213
Table 6.11 Five principal components representing linear combinations of 18 habitat characteristics and the variance explained by each. 215
Table 6.12 Spearman rank correlations between native species richness ranking, maximum rarity ranking, NMDS axes and five principal components. 218
Table 6.13 Grazing-sensitive species (SIMPER), their contribution to Bray–Curtis dissimilarity and abundance at regularly (n = 13) and rarely (n = 9) grazed river red gum and coolibah sites. 220
Table 6.14 Grazing-tolerant species (SIMPER), their contribution to Bray–Curtis dissimilarity and abundance at regularly (n = 13) and rarely (n = 9) grazed river red gum and coolibah sites. 221
Table 7.1 Mean total plant species richness, functional diversity and ecosystem service provision rankings by vegetation type. 253
Table 7.2 Spearman correlation matrix for ecosystem services, total species richness and functional diversity. 254
Table 7.3 Spearman correlation matrix for ecosystem services and biomass of potential ESPs. 254
Table 7.4 Spearman correlation matrix for ecosystem services and cover of potential ESPs. 255
Table 7.5 Five principal components representing linear combinations of 18 habitat characteristics and the variance explained by each. 256
Table 7.6 Grazing impact on ESPs and ecosystem service provision in river red gum and coolibah vegetation types. 257

Chapter1: Introduction

Cotton and native vegetation on the lower Namoi floodplain (February 2005).

Chapter 1: Introduction

1.1 Introduction

Human well-being and biodiversity are inextricably linked (Balvanera et al. 2006; Butler & Oluoch-Kosura 2006; Daily 1997; MA 2005). Ecosystem services provided by biodiversity sustain human societies, but in an attempt to maximise well-being, humankind's dominance over the earth has led to biodiversity loss (Kareiva et al. 2007; Kremen 2005; Kremen & Ostfeld 2005; MA 2005). Increasingly, shifts in and losses of biodiversity are affecting basic life-support functions, though these impacts are not equally distributed, and are felt most by regional populations and, in particular, by the poor (MA 2005). The intensity of urbanisation and extent of agricultural land use are, without doubt, the greatest impacts on biodiversity and ecosystem service provision (Heywood 1995; Jackson et al. 2009; MA 2005; McNeely & Scherr 2003). Hence, better management of these land uses is an obvious starting point in securing future ecosystem service provision and the biodiversity that underpins it.

Society expects that agricultural lands will provide both food and fibre, as well as the ecosystem services required to maintain quality of life (Björklund et al. 1999; McNeely & Scherr 2003; Swift et al. 2004; Swinton et al. 2007; Tscharntke et al. 2005; Zhang et al. 2007). In many cases, this goal has been compromised by inappropriate management and a culture of belief that ecosystem services are provided free of charge (Daily 2000; Proctor et al. 2002). The impacts of changes in biodiversity and ecosystem service provision will be felt hardest by agriculture as the level of costly artificial inputs required to produce goods increases, or technological substitutes fail to adequately carry out the functions of the natural environment (Tscharntke et al. 2005; Kremen et al. 2004, 2007). This will have flow-on effects for the general population as the price of basic goods increases and availability declines.

Much of the arable land of New South Wales (NSW), including the majority of the wheat–sheep zone has been moderately or highly modified by humans since European settlement (Benson 1991; Cox et al. 2001). Outside urban areas, land with fertile soils and flat or undulating topography has been cleared and converted to irrigated or dryland cropping uses, while areas unable to be cropped are extensively grazed by introduced livestock (Benson 1991). The native vegetation of the lower Namoi floodplain has been extensively cleared and modified by agriculture. No areas of the floodplain are part of the formal conservation estate (i.e. the Australian National Reserve System; Commonwealth of Australia 2009). The large majority is privately owned and managed for agriculture. Approximately 10% of the lower Namoi floodplain is under irrigated crops, a further 30% is dryland cropping and another 55% is grazed (predominantly native pasture). Of the remaining 5%, the majority is farm dams or urban areas. Given the significance of the lower Namoi floodplain for agriculture, particularly the intensive irrigated production of cotton, the broad set of values associated with remnant native vegetation (Gillespie et al. 2000) is often not recognised. In order for well informed decision making about the native vegetation estate in this and similarly developed agricultural regions, research is required to quantify the broad set of values associated with remnant natural and semi-natural ecosystems in farming landscapes and the influence of management on those values. Landholders, regional authorities, governments and society will then be in an informed position to make decisions about the future of the remaining natural and semi-natural ecosystems in such regions.

This thesis aims to quantify the provision of three different ecosystem services from the five vegetation communities most widely distributed across the lower Namoi floodplain. The key factors determining vegetation condition or the ability of the vegetation to provide ecosystem services are identified. The effects of management (i.e. grazing intensity) on service provision and hence trade-offs associated with this land use are also ascertained. The three ecosystem services chosen, of the myriad services possible, were carbon sequestration, erosion mitigation and biodiversity conservation. These three services were highlighted, along with natural pest control, salinity mitigation and forage production, as important issues by cotton growers and Namoi catchment management bodies in a planning meeting during the early stages of this project. Cotton farms and public land were chosen to conduct the field work for this research because natural areas on cotton farms cover frequently inundated riparian zones on silty alluvial soils through to sandy, sodic ridges that are rarely flooded, making for the maximum degree of natural variability in remnant vegetation. Management of natural areas on cotton farms is also highly variable, ranging from conservation through to regular sheep or cattle grazing with varying intensities of management. The remainder of this chapter reviews the literature relevant to the concept of ecosystem services and their provision by vegetation, and the relevance of the ecosystem services concept to cotton growers (and therefore farmers and graziers, more generally). Gaps in ecosystem services research are identified, and the aim and objectives of the thesis are detailed. An outline of the rest of the thesis is provided at the conclusion of this chapter.

1.2 Ecosystem Services: The Concept

Ecosystem services are the conditions and processes through which ecosystems and their constituent species sustain and fulfil human life (Daily 1997). They are the benefits that human populations derive directly or indirectly from ecosystem functions (Costanza et al. 1997; MA 2005). A recent description of ecosystem services describes them as the steady flow of 'interest' that humanity derives from 'natural capital' (Greiner et al. 2009). Natural capital is 'an economic metaphor for the stock of physical and biological natural resources that consist of renewable natural capital (living species and ecosystems); non-renewable natural capital (subsoil assets, e.g. petroleum, coal, diamonds); replenishable natural capital (e.g. the atmosphere, potable water, fertile soils); and cultivated natural capital (e.g. crops and forest plantations)' (Aronson et al. 2007). The ecosystem services concept sees human activities as integral components of Earth's ecosystems, focusing on a future where planetary life support systems are maintained while meeting human needs (Palmer et al. 2004).

Declining natural capital (Daily 2000) and a loss of ecosystem services in some areas have sparked a rush of publications on the importance of ecosystem services to human society (de Groot et al. 2002). Attempts have been made to put an economic value on ecosystem services but the methods are much debated as there are no well established markets for most services (Costanza et al. 1998). A pivotal work on the valuation of ecosystem services by Costanza et al. (1997) generated controversy when 17 global ecosystem services were valued at US$33 trillion. While the paper was criticised by many (Opschoor 1998; Serafy 1998; Toman 1998; Turner et al. 1998), it alerted researchers to the importance of ecosystem services, generated debate and escalated research on ecosystem service measurement and valuation.

The concept that ecosystems support humans and that anthropogenic activities impact on the natural environment and sustainability of civilisations is not new. Modern concerns about ecosystem services probably arose from George Perkins Marsh's (1864) publication, *Man and Nature*, and the subsequent 1874 revision, *The Earth as Modified by Human Action: Man and Nature* (Mooney & Ehrlich 1997). The concept of ecosystem services received further attention in the 1970s being referred to as 'environmental services,' 'public-service functions of the global environment,' 'public services of the global ecosystem' and 'nature's services,' before the term 'ecosystem services' was settled on in the early 1980s (Mooney & Ehrlich 1997). The publication of two works, *The Value of the World's Ecosystem Services and Natural Capital* by Costanza et al. (1997) and *Nature's Services: Societal Dependence on Natural Ecosystems* edited by Daily (1997), generated widespread academic and bureaucratic interest in the concept of ecosystem services. The concept gained further momentum with the publication of the *Millennium Ecosystem Assessment* (MA 2003, 2005).

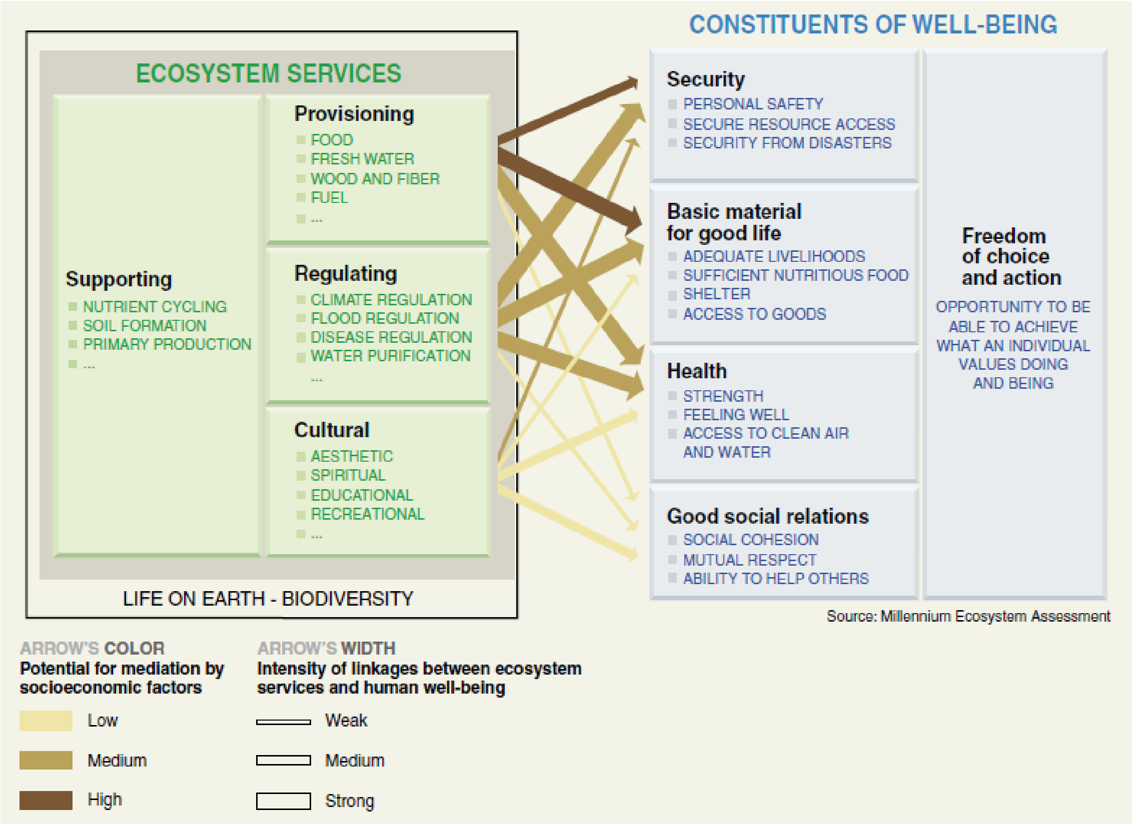
1.3 Why is the Ecosystem Services Concept Topical?

While many natural and semi-natural areas are managed to produce ecosystem goods such as wood and fish, few areas are managed or valued for the services they provide (Kremen et al. 2004). The relationships between ecosystem services are likely to be highly non-linear (Farber et al. 2002) and are often interdependent (MA 2006). This makes their management difficult and trade-offs often arise where attempts to maximise a single ecosystem service occur. The need to produce food and fibre and a poor understanding of the value of natural ecosystems in providing ecosystem services has led to the conversion of large tracts of wildlands across all continents in agricultural lands. A loss of other life-supporting services has occurred to the point that human well-being is now affected in many ways (MA 2005).

Acknowledgement that natural ecosystems have a direct influence on the sustainability of human civilisations and their future prosperity provides new recognition of the need to conserve biodiversity. Past approaches encouraging the conservation of biodiversity have concentrated on the intrinsic value of species and have largely ignored their functional value (Lerdau & Slobodkin 2002). The ecosystem services framework provides a practical and flexible approach to environmental protection that integrates biophysical, economic and social factors and can therefore be understood by a wider audience (Daily 2000). This is of far-reaching importance considering the environmental impacts that will occur in order to sustain a growing global human population (Daily 1997; MA 2005; Tilman 1999).

The MA was undertaken with the aim of exploring future changes in the provision of life-supporting services by the world’s ecosystems (Carpenter et al. 2006) and establishing a scientific basis for enhancing the contribution of ecosystems to human well-being without undermining their long-term productivity (MA 2003). The MA quantified the effects of humans on many of the Earth’s ecosystems and estimated the consequences of resultant losses of ecosystem services on human well-being (Figure 1.1). The results suggest that while changes to ecosystems over the past

50 years have contributed to net gains in human well-being and economic development, the cost of these gains has been the degradation and unsustainable use of up to 60% (15 of 24) of the ecosystem services examined. These gains have not been uniformly distributed across all parts of the world, and have increased the likelihood of abrupt alterations in water quality, disease emergence, collapse of fisheries and shifts in regional climate.



Source: Millennium Ecosystem Assessment

Figure 1.1: Millennium Ecosystem Assessment diagram depicting the strength of linkages between categories of ecosystem services and components of human well-being

1.4 Classification of Ecosystem Services

Classification of ecosystem services has been approached in several ways (de Groot et al. 2002; MA 2005; Wallace 2007). While there is some overlap in classifications used by various authorities, there is no universally agreed classification of ecosystem services. The complexity of ecological functions underpinning ecosystem service provision may militate against a single taxonomy. The classification of various services into groups is often author-specific and so many classifications fail to serve broad global objectives. It is conceivable that there is no one correct method of ecosystem service classification: it depends on the purpose of the classification. The MA (2003) summarised past approaches to ecosystem service classification into three main categories: functional (e.g. de Groot et al. 2002), organisational (e.g. Norberg 1999) and descriptive (e.g. Moberg & Folke 1999) groupings.

The functional approach to ecosystem service classification is the most widely adopted system. De Groot et al. (2002) assigned ecosystem services to regulation, habitat, production and information functions. Daily (1999, 2000) used a similar approach with five categories: production of goods, regeneration processes, stabilising processes, life-fulfilling functions and preservation of options. The MA (2005) also used a functional approach to ecosystem service classification with four categories (provisioning, regulating, supporting and cultural services). In the past, ecosystem services were divided into goods (direct benefits) and services (indirect benefits). However, the authors of the MA (2003) argued that it is often difficult to distinguish between environmental ‘goods’ and ‘services’, and benefits such as cultural values and other intangible services are sometimes forgotten under this approach.

Norberg (1999) developed an organisational approach to classify ecosystem services but it has not been widely adopted. Norberg’s classification was based on three questions: (1) Are the goods or the object of the service internal to the ecosystem? (2) Are the goods or object of the service of biotic or abiotic origin? (3) At what level of ecological hierarchy are the goods or services maintained? Three categories of ecosystem service emerge when these criteria are applied: (1) those associated with certain species or a group of similar species, (2) those that regulate exogenous chemical or physical inputs and are reliant on the entire community or ecosystem, and (3) those related to the organisation of biotic entities such as gene sequences through to networks of energy and material flows. Norberg’s classification draws heavily on ecological theory and therefore does not include services of a social nature classified by other authors as cultural services. Norberg’s approach may therefore be considered incomplete.

Moberg and Folke (1999) classified ecosystem services of coral reefs using a descriptive approach based on how they are generated. Goods and services were classified separately, goods being classified as renewable or reef mining (non-renewable) and services being classified as physical structure services, biotic services, biogeochemical services, information services and social and cultural services.

The MA acknowledged that ‘supporting’ services are necessary for the production of all other ecosystem services, implicitly recognising a hierarchical scale of importance of ecosystem services. Earlier work by de Groot et al. (2002) suggested that ‘regulation’ and ‘habitat’ functions are essential to the provision of ‘production’ and ‘information’ services. The grouping of individual ecosystem services by de Groot et al. (2002) and the MA (2005) into functional categories was approached differently resulting in ‘regulation’ services (de Groot et al. 2002) being split into ‘regulation’ and ‘supporting’ services in the MA classification and ‘habitat functions’ (de Groot et al. 2002) becoming ‘supporting’ services in the MA classification.

Since the publication of the MA, Wallace (2007) has pointed out that many of the ecosystem services currently included in inventories (such as that used in the MA), are processes that lead to the provision of services and are not ecosystem services as such; many commonly used schemes mix ‘means’ with ‘ends.’ In his classification of ecosystem services, Wallace used the categories ‘adequate resources,’ ‘benign physical and chemical environment,’ ‘protection from predators, disease and parasites,’ and ‘socio-cultural fulfilment’ to group services.

1.5 Vegetation as an Ecosystem Service Provider

In addition to moral and ethical arguments for biodiversity conservation, human well-being depends on ecosystem services provided by biodiversity (MA 2005). Many services (such as pollination, pest control and nutrient cycling) are pertinent to the future sustainability of agricultural landscapes and operate at local to sub-catchment scales (Tscharntke et al. 2005; Zhang et al. 2007). Other ecosystem services provided by vegetation remnants (such as climate regulation, carbon sequestration and erosion mitigation) have wider societal value as their benefits extend beyond the farm boundary.

In addition to their ecological importance and value in providing regulating and supporting services, vegetation remnants have economic and cultural importance. Much of the remnant vegetation across Australia is grazed, providing income for landholders and food and fibre for people. Examples of other provisioning services derived from native vegetation include production of timber, bush foods and honey, medicinal products, cut flowers and jewellery. In terms of social values, native vegetation provides opportunities for recreation, education and ecotourism, aesthetic value and in many cultures, spiritual and historical values (Daily 1997). There are both synergies and trade-offs with management for biodiversity conservation and provision of ecosystem services: not all ecosystem service provision is compatible with biodiversity conservation (Turner et al. 2007). Land occupied by native vegetation may also be valuable for development of infrastructure and crop production; hence the opportunity costs associated with conservation of remnants must be weighed up by the landholder and society.

Table 1.1 shows vegetation is directly or indirectly a provider of many ecosystem services. However, not all services are provided by all vegetation types. For example, open savannah woodland may be ideal for forage production, but is poor habitat for fauna requiring dense tree or shrub cover and hence may not be optimal for habitat provision (Cunningham 1976; Parkes & Lyon 2006). It is of only minor value for carbon sequestration where large amounts of woody biomass are required for maximum carbon storage (Young et al. 2005). At the same time, a single vegetation community has multiple values but exploitation for one value may preclude another (Turner et al. 1998). Within a forest, for example, provision of a service such as recreation may limit or preclude other services such as timber production. In order to understand why some ecosystems are better at providing certain services when compared to other ecosystems, or why some services are precluded when other services are exploited, an examination of the concept of ecosystem service providers (ESPs) is necessary.

Table 1.1: Ecosystem services provided by terrestrial ecosystems (Reid 2010).

I. Provisioning services

1. Food and beverages
2. Forage
3. Fibre
4. Timber and wood products
5. Biomass fuel
6. Genetic resources
7. Natural biochemicals
8. Ornamental resources
9. Fresh water
10. Raw materials for manufactured products

II. Regulating services

11. UVb protection
12. Air purification and gas regulation
13. Climate regulation
14. Carbon sequestration
15. Nitrogen fixation
16. Surface water eco-regulation
17. Groundwater eco-regulation
18. Water purification and waste treatment
19. Water transportation
20. Maintenance of soil health
21. Erosion control
22. Terrestrial waste absorption and breakdown
23. Habitat provision
24. Eco-regulation of human disease
25. Biological and natural pest control
26. Resistance to invasion by pests
27. Biotic pollination
28. Dispersal and nutrient translocation
29. Provision of shade and shelter
30. Firebreak and fire hazard mitigation
31. Noise mitigation
32. Coastal storm protection
33. Ecosystem stability and resilience

III. Cultural services

34. Cultural identity and diversity
35. Spiritual and religious values
36. Knowledge systems (traditional and formal)
37. Educational values
38. Inspiration
39. Aesthetic values
40. Social relations
41. Psychological health and wellbeing
42. Sense of place
43. Cultural heritage conservation
44. Natural heritage and biodiversity conservation
45. Recreation and tourism
46. Existence value
47. Option value
48. Bequest value
49. Land value

IV. Supporting Services

50. Primary production
51. Secondary production
52. Production of atmospheric oxygen
53. Soil formation
54. Nutrient cycling
55. Water cycling
56. Maintenance of biodiversity
57. Reproduction
58. Evolution
59. Ecosystem dynamics and succession

Ecosystem service providers are the component populations, species, functional groups (guilds), food webs or habitat types that collectively produce ecosystem services (Kremen 2005). The contribution or functional importance of each ESP to service provision is a product of its efficacy in providing the service, and its abundance in the ecosystem (Balvanera et al. 2005). A related concept, the service providing unit (SPU: Luck et al. 2003) focuses on the characteristics of organisms and populations required to provide ecosystem services. The SPU and ESP concepts were combined in 2009 to form an SPU–ESP ‘service provider’ continuum (Luck et al. 2009). The SPU–ESP continuum promoted the quantification of organism, community or habitat characteristics required to provide ecosystem services in light of beneficiary demands and ecosystem dynamics (Luck et al. 2009).

To assess the effect of management on ESPs, Kremen (2005) articulated two complementary approaches: an ESP-centred approach investigates factors affecting the distribution, abundance or efficiency of individual ESPs to facilitate management specifically targeted at populations of that ESP. The function-centred approach uses multivariate methods to identify environmental factors influencing aggregate function and its variability in order to identify management priorities to maximise ecosystem service provision.

1.6 Vegetation Condition

Vegetation condition is context-dependent: condition depends on the intended use of land (Dudley et al. 2006; Oliver et al. 2002; Tongway & Ludwig 1997), so there is no standard definition of vegetation condition (Gibbons et al. 2006; Gibbons & Freudenberger 2006). However, broadly speaking, vegetation condition is defined as the capacity of a plant community to provide goods and services (Thackway & Lesslie 2006). In an ecosystem service context, this definition can be refined to describe the capacity of vegetation to provide particular benefits desired by humans.

Approaches to vegetation condition assessment predominantly fall into three categories, condition defined in terms of: (1) productive potential, usually in an agricultural sense (e.g. for grazing; Steenekamp & Bosch 1995) or for timber production (Fox 2000), (2) vegetation functionality, such as the ability of vegetation to prevent erosion and retain resources (Ludwig et al. 1997), and (3) current state compared to a baseline state (e.g. in Australia, a pre-European state is often used to assess vegetation condition for biodiversity conservation; Gibbons et al. 2008; Hopkins 1999). The choice of which approach to take is governed by the value systems and purpose of those assessing vegetation condition, thus variation in both concepts and methods has emerged.

The notion of vegetation condition assessment was developed in the US rangelands in the early 1900s with the application of the Clementsian concept of plant succession and climax to range condition assessment and management (e.g. Clements 1916; Sampson 1917). Range condition and trend assessment methods are now generally based on the state and transition framework (Westoby et al. 1989), which encompasses both the early concepts of single equilibrium communities and deterministic successional pathways as well as the later recognition of alternative stable states, discontinuous and irreversible transitions, non-equilibrium communities and stochastic effects in succession.

An alternative view of vegetation condition assessment comes from the rangelands of Australia and focuses on measurement of functional integrity or the ‘intactness of soil and native vegetation patterns and the processes that maintain these patterns’ (Ludwig et al. 2004). Vegetation condition in this case is based on a landscape’s ability to retain resources vital for primary production and other ecosystem responses (Bastin & Ludwig 2006). Landscapes in poor condition are considered ‘leaky’ when these resources cannot be contained and there is a net and continuing loss from the system, whereas competent landscapes in good condition are retentive of water and nutrients (Ludwig et al. 1997).

The Landscape Function Analysis methodology (Tongway & Hindley 2004) used to measure landscape ‘leakiness’ recognises the significance of vegetation cover and its configuration as well as soil surface condition in determining landscape functionality.

Vegetation condition assessment in forestry has, until recently, emphasised the productive potential of a site or the ‘yield capacity of a piece of land according to the nature of the soil and the climate to which it is subjected’ (Schlich 1896). Site quality was determined by a site index, defined by the average height of dominant (or dominant and co-dominant) trees at a specific reference age (Wang 1998). However, following the 1992 Rio Earth Summit and the subsequent development of the Montréal Process and signing of the Santiago declaration, other forest values have been acknowledged and many of these are now incorporated into forest condition assessments (Anon. 1995).

The most recent application of vegetation condition assessment has been for biodiversity conservation, particularly in Australia (e.g. Habitat Hectares: DSE 2004, Biodiversity Benefits Index: Oliver et al. 2005, Biometric: Gibbons et al. 2008, BioCondition: Eyre et al. 2006, and Bushland Condition Monitoring: Croft et al. 2009) and the USA (e.g. Andreasen et al. 2001; Ribaudo et al. 2001). For assessing biodiversity values, a combination of structural, functional and compositional indicators is used to assess vegetation condition at the site or patch scale. Attributes may be compared to benchmark values derived from a long undisturbed stand or the best example of that vegetation type left in the landscape (Parkes et al. 2003; Ayres et al. 2005). Landscape context and connectivity as a measure of species dispersal and persistence in the landscape are also considered important attributes in biodiversity conservation value assessments (Oliver et al. 2007).

In Australia, which was settled by Europeans beginning in 1788, pre-European benchmarks are typically used to compare extant vegetation with that which might have existed prior to changes brought about by, for example, the introduction of exotic species, land clearing, alteration of flood regimes, grazing by introduced livestock and altered fire regimes. Arguments have been raised over the use of pre-European benchmarks. Oliver et al. (2002) stated that use of pre-European benchmarks devalues (in terms of biodiversity conservation) native vegetation that differs in type from that thought to have existed pre-1750. In agriculture-dominated catchments, in particular, any remaining vegetation, especially riparian vegetation, provides habitat and refuge for biodiversity and provides essential ecosystem services. Thoms et al. (1999) and Palmer et al. (1997) argued that in extensively modified environments, it is often not economically possible to return ecosystems to their pre-development state; unrealistic reference conditions based solely on pre-development (pre-European) conditions should not form the platform for future management. Oliver et al. (2002) suggested a better assessment of condition is obtained by assessing the value of what exists now, rather than what once existed.

Another recent application of vegetation condition assessment is in assessing the success of restoration or rehabilitation efforts. This is usually after cessation of some form of land use, in particular mining or agriculture. Indicators of vegetation condition or restoration success are determined by the goals of restoration, the ultimate goal being to create a self-supporting ecosystem that is resilient to perturbation without further assistance (Ruiz-Jaen & Aide 2005).

1.7 The Australian Cotton Industry

Cotton (*Gossypium hirsutum*) has been produced in Australia for fibre and oil since the early 1920s (Pratley 2003). Earlier attempts to grow cotton in Australia began with the arrival of seeds with the first fleet in 1788 (Basinski 1963), but unfavourable climate, soil and seed varieties, and the lack of cheap labour hindered large-scale production (McHugh 1996). The modern Australian cotton industry began on the Namoi River in northern NSW with 25 ha planted near Wee Waa by two Californian farmers in 1961 (McHugh 1996). The completion of Keepit dam upstream on the Namoi River soon followed and cotton production expanded to take advantage of the available water resource. Cotton growing spread in NSW to the Gwydir Valley with the completion of Copeton Dam in 1976, the Macintyre River Valley (Glenlyon Dam) in 1977 (McHugh 1996) and the Macquarie Valley in 1966 (Burrendong Dam) (Arthington 1996). Cotton is also grown in Queensland at Emerald, St George and on the Darling Downs, and in the Ord River Basin in Western Australia.

The area planted to cotton in the Namoi Valley averages 90 000–100 000 ha, but this has declined in recent seasons owing to reduced water allocations to irrigation unsatisfactory climatic conditions and lower commodity prices (Cotton Grower Services, pers comm., 5 May 2009). The majority of production occurs on self-mulching, grey and brown cracking clay soils (SPCC 1980; Ward 1999) of the Vertosol order (Isbell 1996) in riparian and floodplain areas of the catchment, with production concentrated between Narrabri and Burren Junction (Figure 1.2). Cotton production in the Namoi catchment totals approximately 15% of cotton production in NSW and had an average value of around $230 million per annum (Namoi CMB 2003) prior to the 2002–2007 drought.

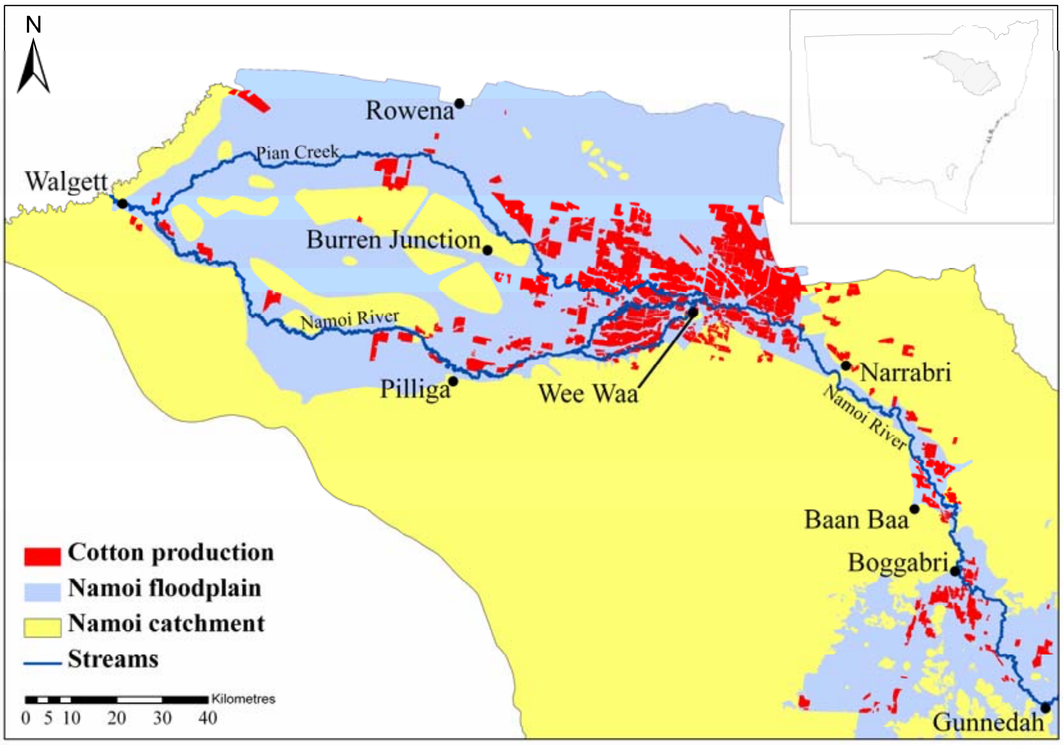


Figure 1.2: Distribution of cotton production in the lower Namoi catchment. Note the concentration of cotton production in riparian and floodplain areas between Narrabri and Burren Junction (data from DLWC 1996).

Most cotton farms are not restricted to cotton production exclusively and it is not uncommon to find sheep and cattle grazing, irrigated cotton and wheat (usually in rotation with cotton) cropping and dryland cropping, on the one property (SPCC 1980). An environmental audit of the cotton industry in 2002 found large areas (>100 ha) of native and regenerating vegetation on 20% of cotton farms, with some farms reporting up to 15% native vegetation cover and one farm planning to have 40% cover at the completion of planned farm development (GHD 2003). On 80% of cotton farms surveyed in the 2002 audit, remnant vegetation occurred in small patches occupying roadsides, areas around houses, adjacent to creeklines, farm borders and areas not viable for cropping (GHD 2003).

The cotton industry in Australia has been aware of environmental issues associated with cotton production for many years (e.g. Cotton Australia 2000; CRDC 2005; GHD 2003; Gibb Environmental Sciences & Arbour International 1991; McHugh 1996; SPCC 1980; Williams 2005). The concerns associated with extensive use of pesticides in particular have been addressed using a variety of approaches including adoption of genetically modified cotton varieties such as Ingard®, Bollgard II® and Roundup Ready®, and encouragement of natural pest control (CRDC 2005). In 1997, the cotton industry introduced the Best Management Practice (BMP) program with a key goal to address the use of chemicals on cotton farms and their impact on the environment (Cotton Australia 2000). This action was prompted by public concern about the use of pesticides in cotton production, traces of cotton chemicals found in livestock, and the impacts of cotton chemicals on surrounding land uses (Cotton Australia 2006; GHD 2003; SPCC 1980).

Since its introduction, the BMP program has expanded to include issues such as application of pesticides, chemical storage and handling, integrated pest management, farm design and management, farm hygiene, storage and handling of petrochemicals, and land and water management (Williams 2005). In 2007, cotton production on BMP-accredited farms was 58% of Upper Namoi, 55% of Lower Namoi (G. Macrae, pers. comm. 9 January 2007) and 18% of Walgett (P. Gibbons, pers. comm. 9 January 2007).

The land and water module of the BMP program was released in 2003 and helped transform BMP into a comprehensive environmental management system (Cotton Australia 2006). This module is yet to attain the same level of acceptance and use by growers as the rest of the BMP program, as there is no driver of comparable magnitude to the chemical issue in the 1990s (Cotton Australia 2006) and cotton production declined about the time of its release. This, coupled with the fact that science has not provided evidence of cause-and-effect mechanisms for some environmental issues or best-practice approaches to ensure desired outcomes, is creating uncertainty and apathy among growers in terms of adoption (Cotton Australia 2006). Lack of availability and access to key resources (e.g. expert advice and technical information) required to execute the land and water management module has also hindered its implementation (Cotton Australia 2006). Basic biological and biophysical data including vegetation mapping, information on species distribution and vegetation community composition is required in order for growers to implement the land and water module.

1.8 Ecosystem Services and Cotton Production

A review of biodiversity research in the cotton industry by Reid et al. (2003) identified 11 ecosystem services relevant to cotton production in eastern Australia. The services identified in the review (natural pest control, maintenance of soil health, prevention of soil erosion, water filtration, waste absorption and breakdown, maintenance of river flows, maintenance of groundwater levels and groundwater quality, maintenance and regeneration of habitat, maintenance and provision of genetic resources, regulation of climate and provision of shade, shelter and barrier effect) were either inputs to production or affected by cotton production. A twelfth ecosystem service, pollination was later added to the list (Reid et al. 2006). Reid et al. (2003) concluded that biodiversity, through the provision of ecosystem services, underpins irrigated and dryland cotton production.

The provision of a carbon sequestration service is of particular interest to cotton growers given the predicted impacts of climate change and the implications it has for agriculture (McRae et al. 2007; Stokes & Howden 2008, 2010). The Australian cotton industry is reducing carbon emissions and taking steps in preparation for the introduction of a future Australian carbon pollution reduction scheme (Bange et al. 2010; Cotton Australia 2009). The introduction of carbon trading markets in the future may see incentives for growers to retain and manage native vegetation for carbon sequestration.

Erosion in cotton growing regions has been considered by some as a minor concern given the relatively flat nature of landscape (Silburn et al. 1999). However, given the variability in groundcover through the year in cotton growing regions, the sodic nature of the soils in many cotton growing regions and the potential for large areas of land to be inundated for long periods of time during flood events, there is potential for extensive and destructive soil erosion. Erosion and soil loss on cotton farms results in reduced water quality due to sedimentation and eutrophication, and the need for higher nutrient and fertiliser inputs, as well as restorative engineering work to clear channels and other irrigation infrastructure, which is costly (Junor et al. 1979; Pimentel et al. 1995). Slumping of river banks near pump sites can result in damage to or loss of infrastructure, again at great cost to the grower (Prosser et al. 2001). Native vegetation in the landscape can provide an erosion mitigation service by capturing sediment and filtering flowing water with suspended sediments, reducing soil loss and stabilising river banks, thereby protecting cotton irrigation infrastructure (Hairsine 2001).

Some minimum configuration of biodiversity is required to provide ecosystem services, many of which underpin and sustain agricultural production. Biodiversity also has cultural values, and many landholders derive satisfaction from biodiversity, for example rare birds or plants on their property (McNeely & Scherr 2003). Biodiversity is required for the day-to-day functioning of the wider landscape, which ensures the sustainability of agricultural activities. Loss of biodiversity has a direct impact on human well-being and is also morally and ethically unacceptable to parts of the wider community (Turner et al. 2003), hence loss of biodiversity may impact growers' 'social licence to farm' (McNeely & Scherr 2003).

1.9 Research Gaps and Thesis Aims

Eamus et al. (2005) posed a series of questions to be addressed by future research that are relevant to this thesis:

1. Which ecosystems or components are best at providing which ecosystem services?
2. What are the impacts of human activities on ecosystem services at local, regional and global scales?
3. What is the relationship between ecosystem services and ecosystem condition?
4. What is required to restore ecosystem services in degraded systems and how would we know when this has been achieved?

5. How interdependent are ecosystem services within an ecosystem and across ecosystems?
6. How much of ecosystem structure and function can be lost before ecosystem services are irretrievably lost?

These questions summarise the gaps that are yet to be answered for most vegetation types and regions of Australia and indeed the world. In particular, little quantitative information exists on ecosystem service provision by native vegetation in inland, northern NSW and the impacts of vegetation condition and management on the provision of ecosystem services. Further, there is scant knowledge of the bulk of the diversity of native species, the ecosystems they form or the services they provide that sustain cotton production and, more generally, cotton farms and regions (Reid et al. 2003). Even for taxa such as higher plants and vertebrates, where systematic surveys have been undertaken of regions in which cotton production occurs, little information has been collected on private land so there are knowledge gaps about the regional ecosystems and species that characterise (or once characterised) the riparian, floodplain, alluvial and wetland landforms and habitats now occupied by cotton farms (Reid et al. 2003). This information is required to form a basis for economic valuation of different vegetation types for cost–benefit analyses of future activities with the potential to impact on vegetation condition. As the Namoi Valley is an agriculturally productive region, retention of large areas of vegetation on fertile soils may entail an opportunity cost (Balvanera et al. 2001). The ecosystem services concept and economic valuation of natural capital provides a more comprehensive assessment of the benefits and costs of native vegetation conversion or modification and greater awareness of the extent of human reliance and well-being on the total environment.

The objectives of this thesis are therefore to:

- Describe the floodplain environment and determine the major environmental influences driving distribution of native vegetation types across the region,
- Map woody vegetation across the study region,
- Quantify the carbon store and carbon sequestration service of the six common native vegetation types of the lower Namoi floodplain,
- Investigate the influence of vegetation type and functioning on aggregate stability and provision of an erosion mitigation service in vertosol soils,
- Assess the biodiversity (vascular plant and bird) conservation value of different vegetation types in diverse condition states, and
- Investigate congruence between plant species and functional richness and diversity on the one hand, and the provision of carbon storage, erosion mitigation and bird conservation services on the other, identify ESPs for each service and the impact of grazing on these services and vascular plant conservation value.

1.10 Thesis Outline

The remainder of this thesis is set out in the following fashion:

- Chapter 2 depicts the study area, the dominant vegetation types of the lower Namoi floodplain and factors pertaining to their distribution in the landscape, land-use history and status in the Namoi catchment,

- Chapter 3 investigated the utility of single-date SPOT5 satellite imagery for mapping vegetation communities across the lower Namoi floodplain,
- Chapter 4 reports on the carbon sequestration value of Namoi floodplain vegetation types,
- Chapter 5 describes the erosion mitigation value of Namoi floodplain vegetation types,
- Chapter 6 presents information on the value of vegetation on the lower Namoi floodplain for vascular plant and bird conservation and the impacts of grazing on this value,
- Chapter 7 investigates the contribution of species and functional richness to the provision of each ecosystem service (carbon sequestration, erosion mitigation and biodiversity conservation), the ecosystem service providers pertinent to ecosystem service provision and trade-offs in ecosystem service provision with increasing grazing intensity, and
- Chapter 8 provides a synthesis of the findings of this thesis, describes research limitations and how this research contributes to current theory. From this, the contribution of the research to resource management practice is described, and vegetation management strategies to maximise ecosystem service provision are recommended. Possible future research on this topic is also discussed.

Chapter 2: What Drives Landscape Scale Vegetation Patterns on the Fertile Semi-arid Floodplains of Northern NSW?

A sharp boundary between Mitchell grass (*Astrebla lappacea*[1]) grassland and coolibah (*Eucalyptus coolabah*) open woodland, delineated by flood frequency and duration of inundation (December 2007).

---

[1] Plant nomenclature in this thesis follows Harden et al. (1993–2002) and for grasses, Jacobs et al. (2008), except where otherwise indicated.

Chapter 2: What Drives Landscape Scale Vegetation Patterns on the Fertile Semi-arid Floodplains of Northern NSW?

2.1 Introduction

Australian vegetation communities are influenced by climate, soil type and topography among other environmental factors (Beadle 1981; Keith 2004; Moore 1970). Within vegetation communities, variation in composition and structure exists due to multiple environmental gradients (e.g. Prober & Thiele 2004). The mechanisms controlling the availability and movement of resources (such as soil type, fertility, and likelihood of flooding), and especially of soil moisture in semi-arid environments, have a strong influence on vegetation community composition, structure and distribution patterns across a range of spatial and temporal scales (Ludwig et al. 1997). Knowledge of these mechanisms can inform management and restoration decisions for different outcomes, including ecosystem service provision.

The semi-arid floodplains of northern NSW consist of some of the most fertile soils in Australia, which, along with open grassy plains, flat topography, available surface and ground water resources and irrigation potential, make them attractive for agriculture (Crapper et al. 1999; Ferry 1978; Stannard & Kelly 1977). As a result, much of the native vegetation has been cleared or substantially altered (Cox et al. 2001; Glanznig 1995; Sivertsen 1994). Remnant vegetation is now protected by legislation pertaining to native vegetation protection and threatened or endangered ecological communities based on threats to long-term survival.

The influence of European development has changed disturbance regimes on the Namoi floodplain. Development for irrigated agriculture has led to changes in flood patterns, and nutrient distribution and movement across the floodplain, affecting species recruitment and soil fertility (Freudenberger 1998). Grazing by sheep and cattle has influenced the composition of remaining vegetation in many areas, especially where palatable species have declined or been eliminated and unpalatable species have increased (Beadle 1948; Grant 2005; Kearle et al. 2002). Grazing by sheep and cattle has also altered carbon and nutrient dynamics (Wilson 1990). Orr (1980a, b) showed that increasing grazing pressure in Mitchell grass grasslands reduced projected foliage cover and basal area of grass tussocks, resulted in the elimination of some perennial grass species and encouraged colonisation of unpalatable annual species. At a finer scale, the introduction of hooved animals has impacted on soil micro-topography and, in some areas (depending on the soil type), led to soil compaction and increased runoff (Greenwood & McKenzie 2001). Introduced weeds such as lippia (*Phyla canescens*) suppress native species because they are superior competitors for resources (e.g. water) (Mawhinney 2003; McCosker 1994) or through allelopathy (McCosker 1994). Members of the Brassicaceae (e.g. turnip weed *Rapistrum rugosum* and London rocket *Sisymbrium irio*) are thought to have changed soil flora and faunal communities, especially mycorrhizal communities (with which many native species form symbiotic relationships), through production of isothiocyanates (antifungal, antibiotic, insecticidal and allelopathic substances) (Annapurna et al. 1996). They are also effective competitors for resources (Campbell et al. 1996).

No broad-scale vegetation survey has been conducted on private land across the lower Namoi floodplain, although Sivertsen and Metcalf (2001) surveyed vegetation on public lands such as Travelling Stock Routes (TSRs) and road reserves in the region. Public land represents a small proportion of the region, is often restricted to agriculturally unproductive land, and experiences different management to privately owned land (Davidson et al. 2005). Hence an accurate picture of regional vegetation patterns cannot be gained from such studies. Remote sensing and modelling studies have provided data on the extent of remnant native vegetation in the region (e.g. Peasley 2000), but these methods cannot provide data on community composition. Overall, vegetation survey effort across the study region has been poor (Eco Logical 2006), with only 10% of all floristic plots surveyed in the region being on private land (Bedward & Ellis 2009). Given the recent political history and sensitivities associated with native vegetation management, this situation is unsurprising.

Few studies have related vegetation community composition to environmental attributes in the study region. Most information about the drivers of distribution patterns for different vegetation types has been extrapolated from studies in adjacent catchments. Where studies have linked vegetation and environmental variables, vague descriptions such as 'red brown earths of rarely flooded backplains' are given. Early studies such as Stannard and Kelly (1977) described landscape units in detail and related the dominant vegetation to them. The best classification of vegetation communities in the study region, to date, is that of Benson (2006). Benson's (2006) comprehensive community profiles apply to a large area of western NSW, and are necessarily broad in interpretation and not a substitute for catchment-scale studies.

2.1.1 Aims and Objectives

The aim of this study was to describe the lower Namoi floodplain environment and to classify the dominant vegetation types across the region according to species composition and structure. The major environmental influences driving the distribution of vegetation types across the region were investigated along with the main drivers of the dominant life-forms of each vegetation type.

2.2 Study Region

2.2.1 Location

The Namoi catchment in northern NSW is 42 053 km^2 in size and drains to the west of the Great Dividing Range. The floodplain of the lower Namoi catchment covers an area of 7163 km^2 extending north and west from Boggabri in the east, to Walgett in the west (Figure 2.1). South-east of Boggabri, the Namoi floodplain is known as the 'Liverpool Plains' but this region is not considered further here. The southern margin of the floodplain west of Narrabri is delineated by the boundary of the cracking clay floodplain soils with the coarser textured Pilliga outwash soils. The northern boundary of the floodplain is unclear due to the relatively flat nature of the region, and is an area where floodwaters from both the Namoi and Gwydir Rivers mix (Hooper & Duggin 1996; Riley & Taylor 1978; Stannard & Kelly 1977). The floodplain boundary for the purpose of this study extended north of the Namoi catchment boundary recognised by the Namoi CMA (Namoi CMB 2003) into the adjacent Gwydir catchment (Figure 2.1), and was determined by the extent of an average flood with a return interval of at least 50 years (A.Falkenmire, pers. comm., 4 December 07). The western boundary of the floodplain near Walgett is flanked by the Barwon River and its floodplain.

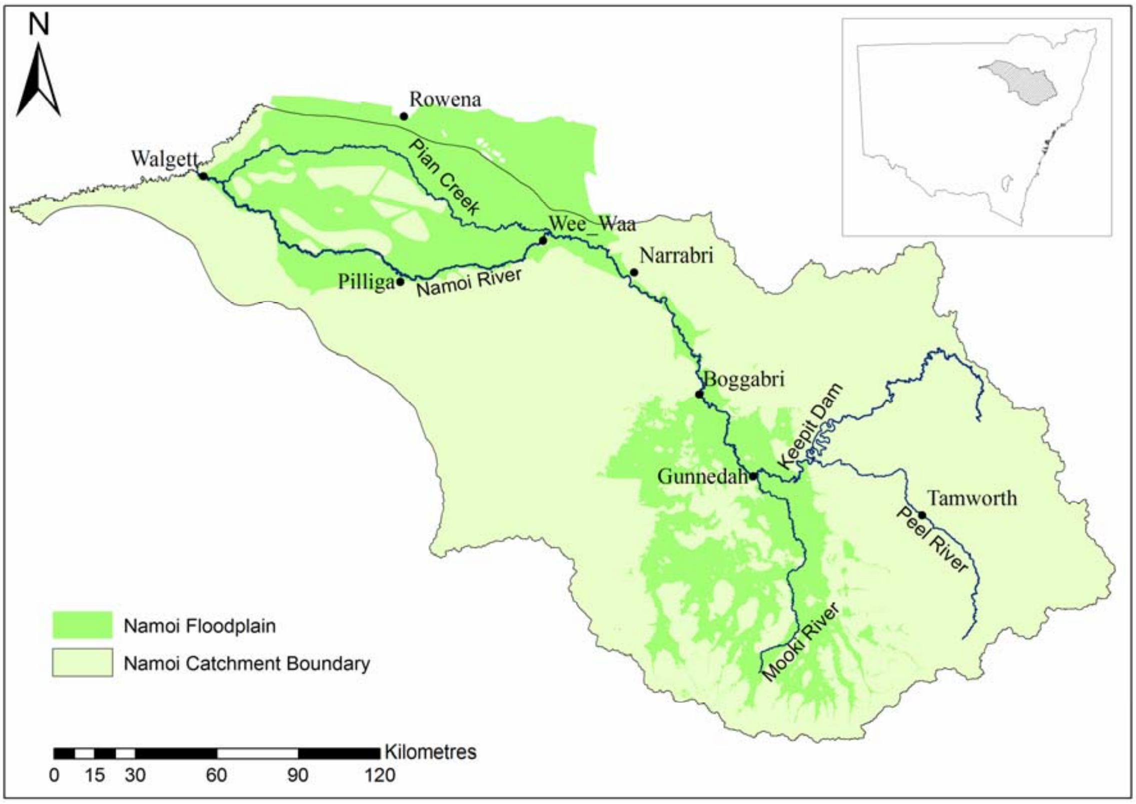


Figure 2.1: Namoi catchment in northern NSW showing the extent of the floodplain. The floodplain between Boggabri in the east and Walgett in the west forms the study region for this study.

The Interim Biogeographic Regionalisation of Australia (Thackway & Cresswell 1995) classifies the continent into bioregions and is based on the dominant variables (climate, landform and soils) determining the occurrence of vegetation across Australia. It shows the Namoi catchment to be dissected by the Darling Riverine Plains, Brigalow Belt South, Nandewar and New England Tablelands bioregions (Figure 2.2), with a minor contribution in the east from the North Coast bioregion (not mapped or discussed further). The lower Namoi floodplain falls predominantly in the Darling Riverine Plains (DRP) bioregion, though sites located just west of Narrabri and between Narrabri and Boggabri fall in the Brigalow Belt South (BBS) bioregion.

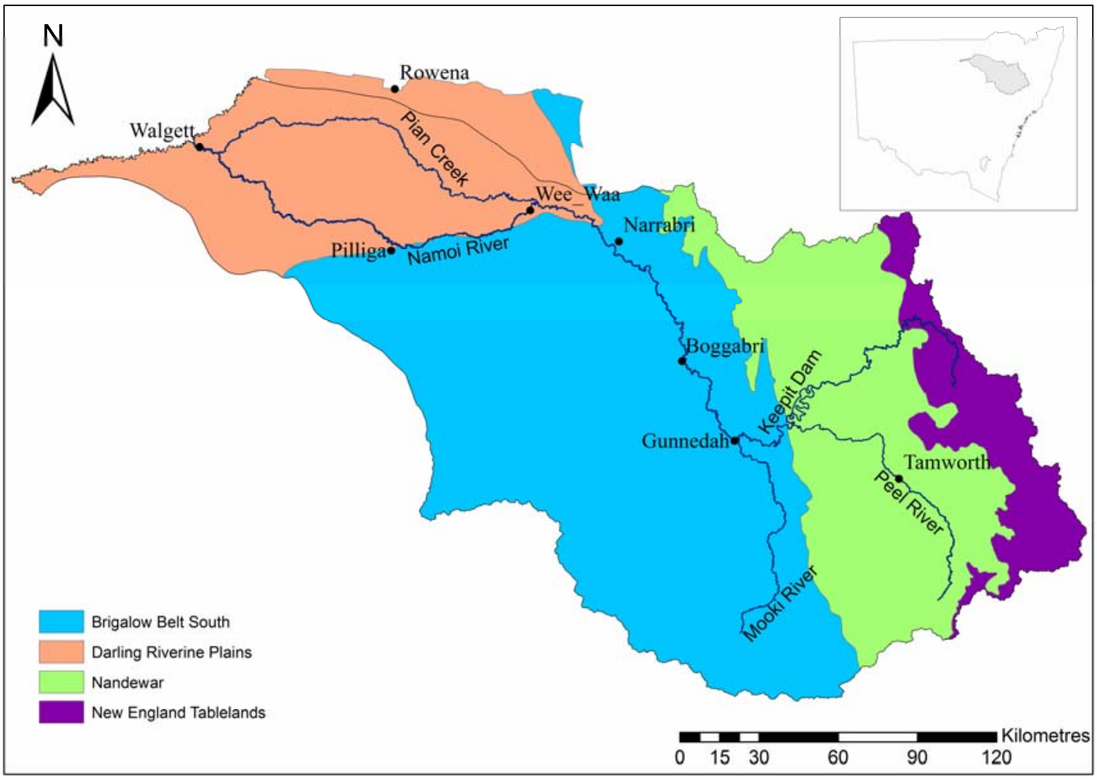


Figure 2.2: The Darling Riverine Plains, Brigalow Belt South, Nandewar and New England Tablelands bioregions dissect the Namoi catchment.

2.2.2 Climate

The Namoi catchment spans the semi-arid, dry sub-tropical and temperate zone boundaries of NSW (Benson 2006). Average rainfall decreases from east to west and varies from >1200 mm in the east to <400 mm in the west (Figure 2.3). The long-term average rainfall of the lower Namoi floodplain is 400 to 600 mm p.a. and has a slight summer dominance (Figure 2.4). The largest falls occur as thunderstorms or as occasional cyclone-related depressions (Kearle et al. 2002; SPCC 1980). Runoff during cyclone-related events can be substantial, though areas in the western portion of the catchment can experience months of little or no rainfall between such events (Kearle et al. 2002; Ward 1999). The spatial and temporal variability of rainfall in the study region indicates the limited value of long-term annual averages in predicting ecological responses and processes such as plant growth and animal breeding patterns

(Kearle et al. 2002; Ward 1999). Mean monthly rainfall and the first and ninth deciles better illustrate rainfall variability across the floodplain (Figure 2.4).

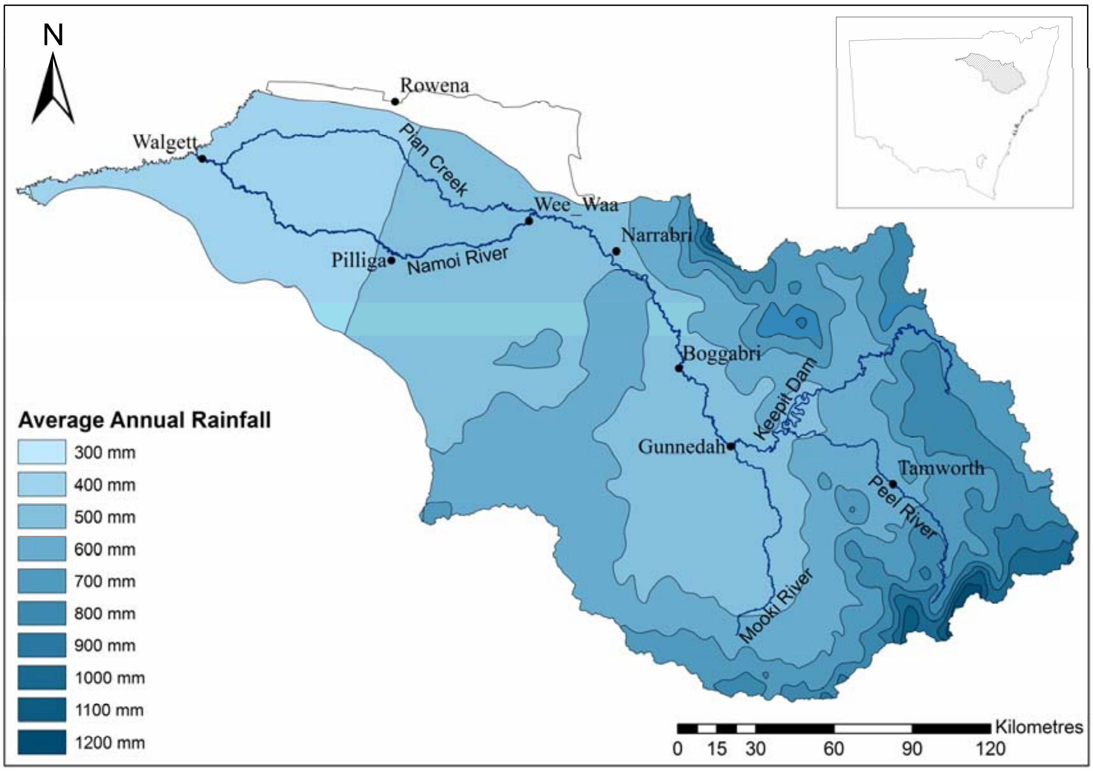


Figure 2.3: Mean annual rainfall across the Namoi catchment.

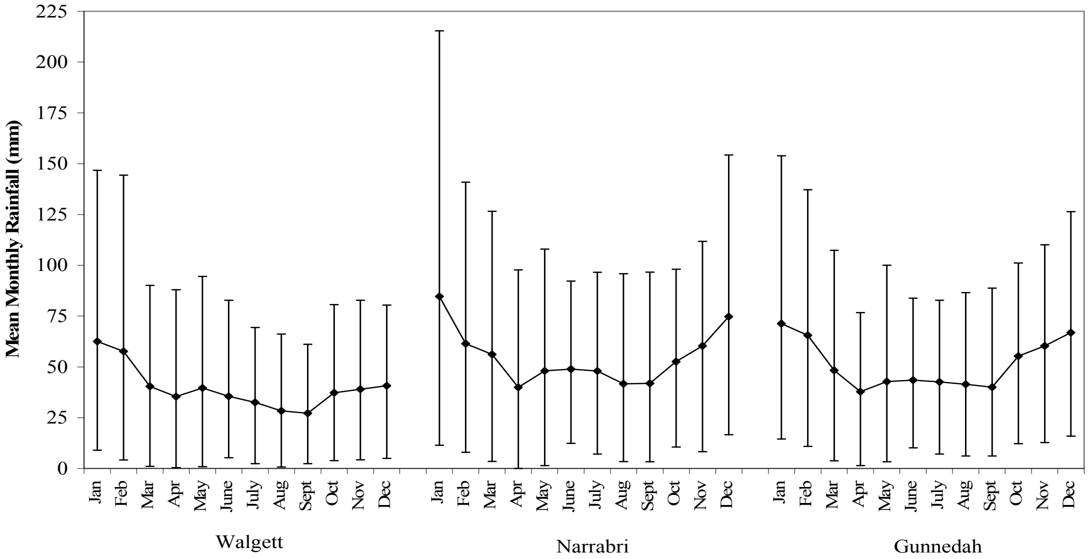


Figure 2.4: Long-term mean monthly rainfall and first and ninth decile data collected at Walgett since 1878, Narrabri 1891 and Gunnedah 1877 (Bureau of Meteorology 2008).

Mean maximum summer temperatures for the lower Namoi floodplain increase slightly from east to west (Figure 2.5) as do mean minimum winter temperatures. Frosts occur in the region, though only a small number of days each year have minimum temperatures below 0ºC (an average of 8.2 days between May and September at Walgett, 11.6 days between May and September at Narrabri and 17.4 days between April and October at Gunnedah) (Bureau of Meteorology 2007).

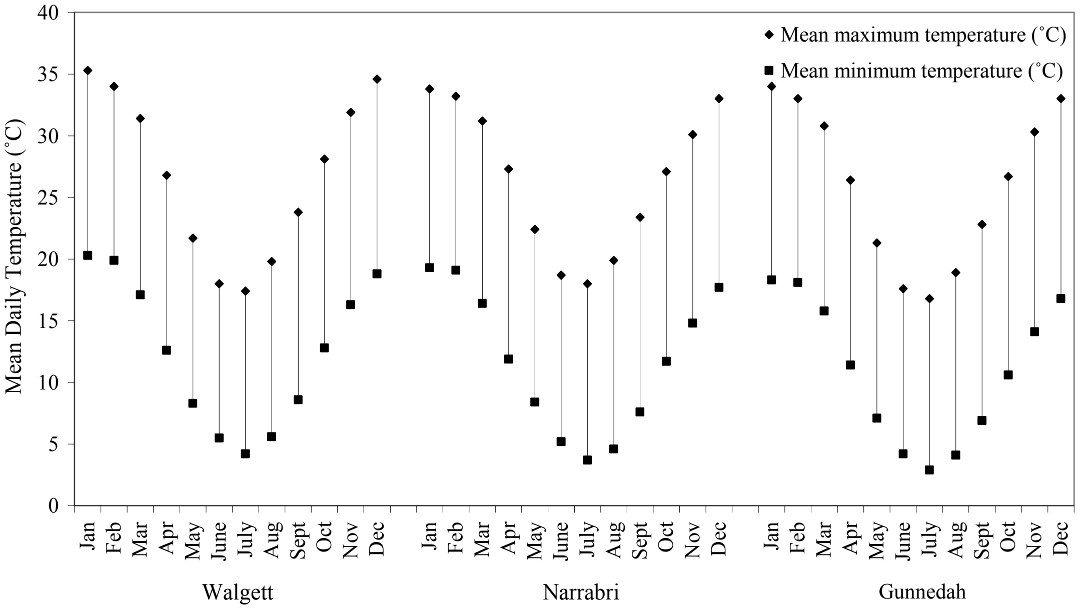


Figure 2.5: Mean daily maximum and minimum temperatures at Walgett (data collected 1878–1993), Narrabri (1962–2002) and Gunnedah (1876–2006) (Bureau of Meteorology 2008).

2.2.3 Landform, Geology, Topography and Drainage

The Namoi catchment can be divided into two regions, the mountainous eastern region and the gently sloping western plains region (Figure 2.6). The western plains consist of unconsolidated sediments derived from alkaline tertiary basalt flows (Stannard & Kelly 1977). The lower Namoi floodplain has a slope less than 1:2000 (SPCC 1980) and is a fluvial environment, shaped by the presence and movement of water across the landscape (Kearle et al. 2002).

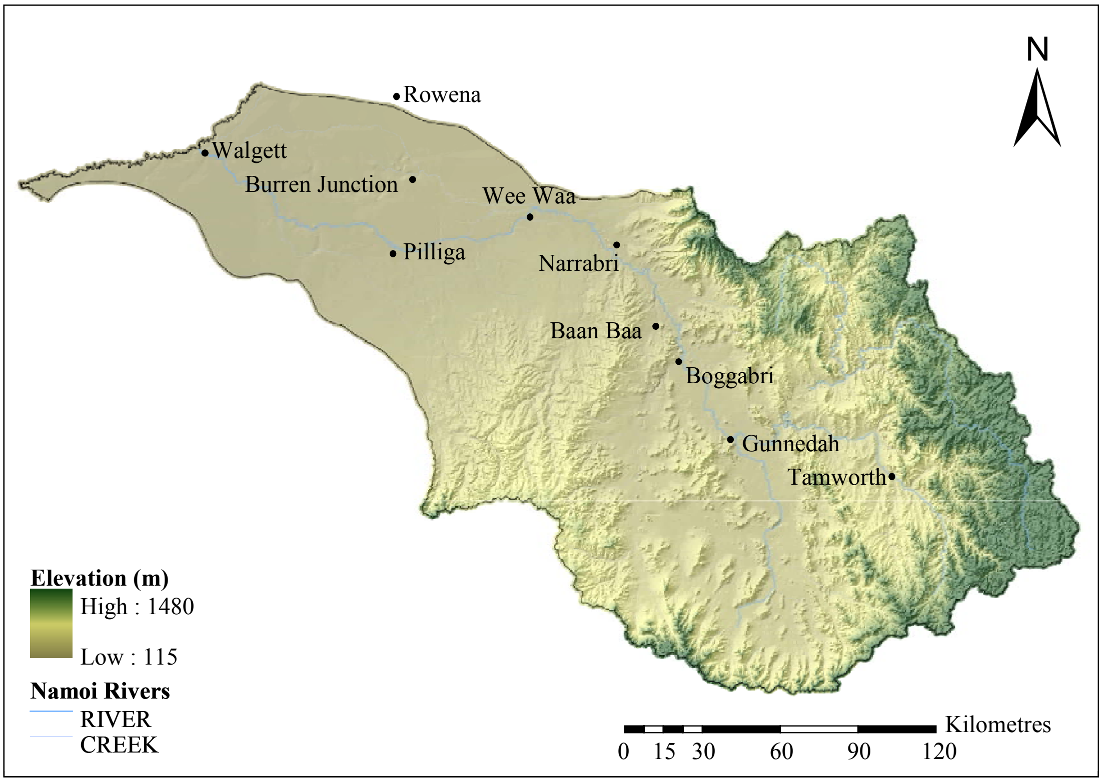


Figure 2.6: Digital Elevation Model of the Namoi catchment showing selected major towns and rivers.

The Namoi River is one of the major westerly flowing tributaries of the Barwon–Darling River system (WRC 1980). The river rises in the Great Dividing Range as the MacDonald River north of Tamworth before becoming the Namoi River and flowing into Keepit Dam, a large storage of approximately 475 000 ML operated principally for the provision and regulation of irrigation water downstream. Other large dams upstream of Keepit Dam (Split Rock Dam, Chaffey Dam and Dungowan Dam) regulate water for irrigation and town water supply on tributaries of the Namoi. The Peel River joins below Keepit Dam while the Mooki River drains the Liverpool Plains and joins the Namoi River at Gunnedah. The Namoi receives most of its inflow before it reaches Narrabri and flows out through the western plains region west of Narrabri. In some rainfall events, a number of small streams drain the Pilliga Scrub and the northern slopes of the Warrumbungle Range to the south.

The southern and eastern perimeter of the Namoi catchment is mountainous; most of the south-eastern region consists of slopes and plains punctuated by hills and outcrops while the western region consists of gently sloping alluvial plains. Irrigation water is in greatest demand during the summer growing season of crops such as cotton, but smaller releases of water from Keepit Dam occur throughout the year to maintain the Namoi River ecosystem. In the past, floodwaters from the Barwon, Macintyre, Gwydir and Namoi Rivers joined to cover an area of approximately 2×10^6 ha between Narrabri, Pilliga, Walgett, Mungindi and Boggabilla (WRC 1980). The last flood of this magnitude was in 1971, with a number of smaller floods since then.

The relatively flat nature of the lower Namoi floodplain encourages the river to become a low-velocity watercourse and has allowed the extensive development of anabranches, billabongs and distributaries (effluent streams) from the main river channel (WRC 1980). The most significant of the distributaries, Pian Creek, diverts floodwater away from the main channel of the Namoi in times of peak flow and has helped form the extensive fertile alluvial plains north of the Namoi River by over-bank deposition of fine alluvium. Pian Creek rejoins the main Namoi River channel east of Walgett before it reaches the Barwon River.

A number of physiographic units can be distinguished within the confines of the lower Namoi floodplain. The main units of interest for the current study east of Narrabri are the alluvial plains, consisting of predominantly black, grey, brown or red clays deposited by the Namoi River and associated tributaries. West of Narrabri, eight units have been distinguished in the lower Namoi valley (Stannard and Kelly 1977), five units being relevant to this study:

1. **Clay plains**–plains dominated by fine-textured dark grey to brown clay soils liable to inundation and of uniform topography except where dissected by stream formations.
2. **Prior stream formations**–are former stream channels, now comprised of belts of slightly elevated and undulating land of coarser texture than the surrounding clay plains. Stream channels within these formations may be lower than the surrounding plains, contain fine-textured sediments, and are preferential paths for floodwaters. Prior stream channels are distinguished from distributaries by shallow channels and wide meander belts, as opposed to the relatively deep channel and narrow meander belt of the latter.
3. **Terminal prior stream floodways**–wide, shallow, broadly meandering channels of grey clay with associated discontinuous levee formations of brown clays. These floodways still carry large volumes of water in floods and represent the downstream part of the prior streams mentioned above where depositional activity was weak or reduced.
4. **Coarse-textured, dissected low floodplain**–a small area associated with the Namoi River between Narrabri and Wee Waa. It is depressed in relation to surrounding plains and is dissected by small flood channels. Coarse-textured sediments characterise the area, which is thought to have been produced through transportation of sediments from the northern margin of the Pilliga Scrub by the Namoi River near Narrabri. As the river passes Narrabri, it runs close to and

dissects prior stream formations and deposits the coarse sediments as a low floodplain. Where low dissected land associated with the river and distributaries is encountered elsewhere in the lower reaches of the Namoi, the sediments are of uniformly fine texture.

5. **Barwon floodplain**–severely dissected by flood channels, depressed in relation to the Namoi clay plains and characterised by both non-self-mulching and self-mulching soils.

2.2.4 Soils

The soils of lower Namoi floodplain are broadly classified as black, grey or brown vertosols (Isbell 1996). Some are sodic, many are calcareous and high levels of Mg are often present owing to the moderately low rainfall and restricted effects of leaching (Ward 1999). The plains consist of a mantle of fine-textured sedimentary material at least 6 m deep, and commonly 9–15 m deep (Stannard & Kelly 1977), grading from a predominantly grey clay to a brown clay below 2 m depth as a result of periods of waterlogging and anaerobic soil conditions affecting the oxidation and reduction of iron (Stannard & Kelly 1977). A sandy clay, sand or gravel layer 1–2 m thick is often encountered below about 5 m depth (closer to the surface near old river courses) but reverts to clay below. The grey clays are predominantly neutral to strongly alkaline and pH increases with depth down the profile. Most of the grey clays are self-mulching and often show subtle gilgai microtopography (Stannard & Kelly 1977). It is believed that they have been deposited recently (in geological terms) by slow-moving floodwaters (McGarity 1950). Low moisture content causes the soil to shrink and crack up to metres deep, but swelling of the clays after rainfall closes the cracks. The shrink–swell action of the clay leads to ‘churning’ of the soil: organic matter and surface soil fall down the cracks as they open up and is incorporated into the subsurface soil as the cracks close. The magnitude of the self-mulching behaviour in the soils of the study region varies from very prominent, where the surface of the soil has a crumb-like structure and large cracks are prominent, to weak. In some areas, no self-mulching characteristic is observed and thin soil crusts are often present. Differences in the behaviour of the surface soil are likely due to differences in clay mineralogy and dominant cations on the exchange complex (Freebairn et al. 1996; Dalal 1989; Dalal & Bridge 1996; Prebble 1987; Yates & McGarity 1984).

Interspersed among the grey clays are areas of brown clay, often in areas slightly higher in elevation than the surrounding floodplain (though the difference is often in the order of centimetres). The brown clays also form extensive plains in the western part of the study region and are generally not subject to regular flooding (Stannard & Kelly 1977). These soils have slightly higher sand content than the grey clays and are often only weakly self-mulching. The colour differences between the grey and brown clays could be the result of much shorter anaerobic periods and better drainage in the brown clays (Stace et al. 1968).

The soils closest to the Namoi River are often lower in clay content, have a lower pH than the surrounding plains and Ca is always the dominant cation (Ward 1999). The lower clay content means these clays do not crack as deeply as the soils of the floodplains. An accumulation of organic matter is often seen in the soil surface horizon as a result of large litter inputs from the highly productive riparian vegetation and lesser self-mulching behaviour of these soils.

2.2.5 Human Occupation, Land Use and Vegetation Management

The lower Namoi floodplain was occupied by the Kamilaroi Aboriginal people prior to the arrival of Europeans and establishment of grazing runs (Ferry 1978; Kass 2003). Very little is known about how the Kamilaroi managed vegetation prior to European settlement, although there is general debate about the use of fire by Aboriginals across Australia (Bowman 1998) and NSW (Benson & Redpath 1997). Following European settlement, traditional management and use of the land by Aboriginals was discouraged, especially where it conflicted with European management objectives, (Kass 2003).

The Liverpool Plains were first seen by Europeans in 1818 when Oxley crossed the region and named the plains and adjacent ranges (Ferry 1978). In 1820, Cunningham visited the region and building on Oxley's earlier reports, included the Liverpool Plains in his list of 'better country further out' (Ferry 1978). Mitchell crossed the Namoi near the present site of Narrabri in 1831 on his expedition north, and spent some time near the Gwydir River and on the lower Gwydir floodplain. Settlers illegally established pastoral enterprises in the area during the 1820s and 1830s (Hunt 1980). Grazing of both sheep and cattle was the dominant land use until the 1880s when large areas were opened up for wheat crops, first on lighter textured soils, then on heavier textured clay soils with the introduction of new technologies in the 1950s (Hunt 1980).

No expedition by explorers across the lower Namoi floodplain has been documented, though Sturt was close to the present day site of Walgett on an expedition across the Castlereagh River in 1828–29 (Ferry 1978). Charles Coxen followed the Namoi River in 1834 along its southern banks as far west as the present site of Pilliga with the objective of collecting natural-history specimens. Coxen gave unfavourable reports of the country upon return, describing it as ‘barren and inhospitable’ (Ferry 1978a). Settlers who took up land along the Namoi River and Pian Creek during the 1830s and 40s were the first recorded Europeans in that area.

A description of the plains in the *Australian Magazine* during the 1840s, in reference to the lower Namoi, Gwydir and Barwon floodplains described the region as ‘scarcely of any wood, and no ridges are to be seen; these plains in the winter season are complete marshes, and fit only for the depasturing of cattle’. Sheep were not considered suited to the area as the plains were deemed ‘too hot and dry in summer for thickly woolled sheep’. Cattle dominated the livestock industry in the region for at least 20 years, as illustrated by the yearly inventories of livestock. During the 1860s, wool prices and cattle disease encouraged a shift to sheep, and the discovery of artesian groundwater in the area in the early 1870s allowed livestock enterprises to expand across the plains away from the Namoi River and Pian Creek. Fencing of paddocks also began at this time.

Drought has influenced land use across the study region, and has subsequently determined pattens of land degradation and vegetation changes through time. Drought was experienced early during settlement, notable dry periods occurring during 1860–

1869 and 1890–1902. During this time degradation of the vegetation occurred due to severe overstocking (Ferry 1978). Reports indicate that after consuming all palatable annual vegetation resources, stock consumed slower growing perennial vegetation and survived largely on saltbush. Inventories of stock numbers indicated many areas were still well-stocked during drought periods, and perhaps overstocked for the conditions. Areas where wells and bores existed reported having plenty of water but no forage, so palatable shrubs such as *Acacia pendula* (myall) were cut for emergency stock feed (Beadle 1948). It was not until the 1910s that graziers started to learn to apply opportunistic grazing strategies: stock was moved in and out of the area according to available pasture.

Ringbarking in the region gained momentum after 1881 and was believed to be an improvement to the land, which encouraging the growth of grasses required for livestock (Ferry 1978). Large areas of *Eucalyptus coolabah* (coolibah) and *Eucalyptus largiflorens* (black box), in particular, were ringbarked near the turn of the century (1900) but, with little follow-up control, many areas regenerated through coppicing of ringbarked trees, creating a more mallee-type growth form with many smaller, spindly stems (Maher 1995). Evidence of ringbarking still exists today with the original ringbarked stems still present on many trees, evidence of the extent of this management activity (Figure 2.7).

Figure 2.7: Coolibah (*Eucalyptus coolabah*) trees at Walgett ringbarked during the early 1900s.

In the 1960s, when new technology such as powerful machinery and large, robust tillage implements allowed cultivation of the heavy clay soils in the region, wheat farming commenced and expanded quickly. Dryland cropping is a dominant land use today. Large areas of land, both grassland and woodland, were cleared to allow the expansion of wheat production. With the completion of Keepit Dam in 1958, irrigated cropping increased in extent, becoming a widespread land use when sufficient water is available

(McHugh 1996). The extent of clearing and cultivation in the region has resulted in a highly fragmented, patchy landscape (frontispiece).

Approximately 25% of the Namoi catchment remains under timber (Mawhinney 2004). The majority of the remnant timber is in the Pilliga Scrub south-west of Narrabri, and in the mountainous ranges in the east of the catchment. The plains of the Namoi catchment have been extensively cleared since European settlement with areas on travelling stock routes (TSRs) or road reserves, steep basalt hills and areas of low agricultural value constituting most of the remaining woody vegetation in arable parts of the catchment (Mawhinney 2004). Large expanses of grassland, with or without scattered trees, still exist interspersed amongst the cropping country and these areas are grazed by sheep and cattle.

2.3 Current Study

2.3.1 Field and Laboratory Methods

Study sites were selected after field reconnaissance, with advice from Namoi Catchment Management Authority staff, and in consultation with private landholders and the Rural Lands Protection Boards (RLPB) in the area. Some 57 sample sites were chosen to represent the dominant vegetation types of the lower Namoi floodplain under the full range of management intensities from little to highly modified. Sites were located on 27 cotton farms, and TSRs managed by the Narrabri and Walgett RLPBs (Figure 2.8). The number of sites on each farm ranged from one to eight, but in some cases multiple farms were owned or managed by the same landholder. Up to six sites were located in the same paddock on many properties as paddock sizes in the area are generally large and cover multiple vegetation communities. Soil type and liability to inundation were not necessarily the same across entire paddocks.

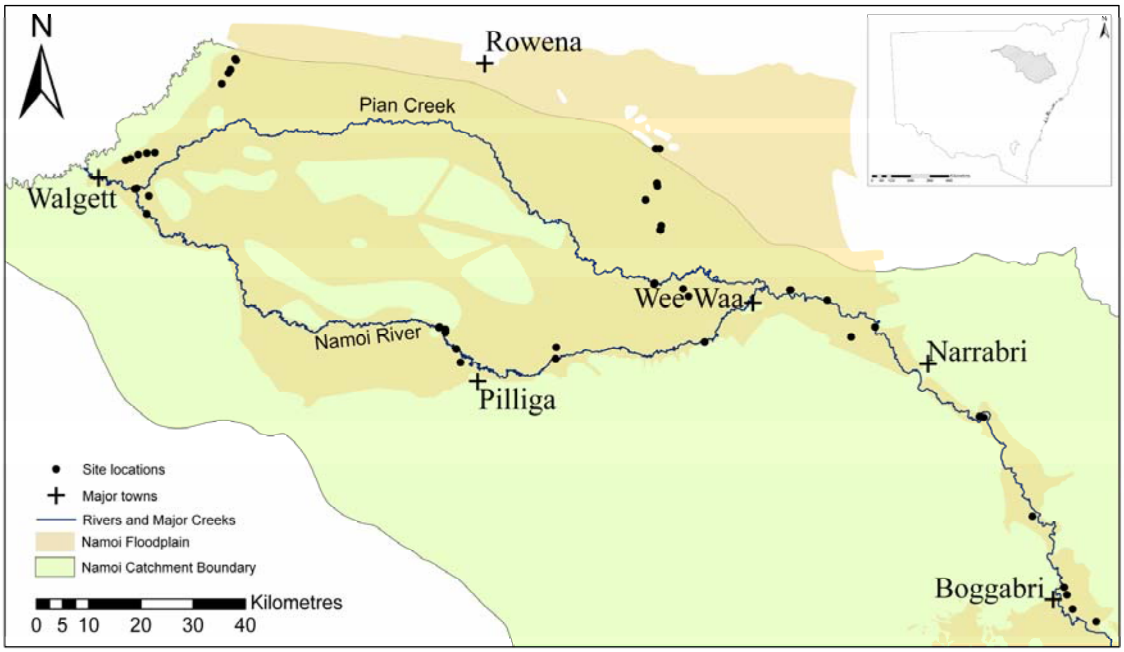


Figure 2.8: Lower Namoi floodplain study region in northern NSW showing location of study sites.

Each site consisted of a 1ha plot in which woody vegetation was sampled, and a 20 × 20 m plot for ground-storey vegetation sampling. For each plot, all vascular plant species and their projected foliage cover were recorded during two sampling periods in April–May 2008 and October–November 2008. Sampling periods occurred in favourable seasons after heavy rain had promoted a strong plant growth response (i.e. maximum number of species and biomass). The native, long-lived perennial component of the vegetation only is reported here (i.e. species native to the study region, and with a life cycle longer than 2 years).

Mean rainfall data for each property was obtained from landholders or, where records were incomplete or unavailable, from the nearest Bureau of Meteorology weather station. Soil cores to a depth of 30 cm were collected at all sites in summer 2007. Nine soil cores were taken using a manual coring device from a 25 × 25 m quadrat centred on the groundstorey vegetation quadrat. Cores were subdivided into depth increments of 0–5 cm and 20–30 cm. Each depth increment was stored in zip-lock plastic bags in cool, shady conditions for up to 1 week during sampling and then at 4°C for up to 2 weeks prior to air-drying.

A bulked soil sample for each depth increment at each site was passed through a 2 mm sieve using a mechanical soil mill after macro-organic matter such as pieces of tree root and leaf were removed. Samples were mixed thoroughly and subsampled for further analyses. Electrical conductivity (EC) and pH were determined in a 1:5 soil to water extract. For soils with pH < 7.5, exchangeable cation (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}) content was determined using atomic absorption spectroscopy following extraction using 1M NH_4Cl. For soils with pH ≥ 7.5, exchangeable cations were determined at the DPI Victoria lab at Werribee using the alcoholic 1M NH_4Cl method (Tucker 1954) after pre-treatment with aqueous ethanol and aqueous glycerol to remove soluble salts. Exchangable cation values were summed to give the effective cation exchange capacity (ECEC) and Ca:Mg ratio was calculated by dividing the Ca content by the Mg content. Exchangeable Ca (ECa), exchangeable Mg (EMg), exchangeable K (EK) and exchangeable Na (ESP) were calculated as a percentage of ECEC.

Soil particle size distribution of samples was determined using the pipette method after overnight end-on-end shaking in $(NaPO_3)_6$ dispersant solution (Day 1965), and expressed as proportions of sand, silt and clay using the international system

(International Society of Soil Science 1929). A subsample of each bulked soil sample was pushed by hand through a 0.5 mm sieve to remove macro-organic matter and sand particles prior to analysis of total C, total organic C (TOC) and N content using a Carlo Erba NA 1500 Solid Sample Analyser.

2.3.2 Statistical Analysis

Many of the soil properties were significantly correlated (Spearman's rank correlation). The final set of environmental variables was exchangeable K at 0–5 cm depth (EK_1), and pH, EC, Ca^{2+}, Mg^{2+}, K^+, Na^+, exchangeable K (EK_2), ECEC, clay, silt and sand fractions (%Clay, %Silt and %Sand, respectively), and N at 20–30 cm depth, and long-term average rainfall. One-way analysis of variance (AOV) (on rank-transformed data where less severe transformations did not satisfy the assumption of normality for parametric AOV). Tukey's HSD pairwise comparisons were used to investigate differences in soil variables between vegetation types using Statistix 8 (Analytical Software 2003).

Sites were classified by dominant canopy species and landscape position in the field. Five vegetation types were recognised: (1) river red gum (*Eucalyptus camaldulensis*) open forests and woodlands of riparian corridors or regularly inundated, low points in the landscape; (2) coolibah woodland and open woodland of lesser drainage lines and the floodplain; (3) myall shrubland and open shrubland; (4) black box woodland; and (5) grassland. Structure was not considered in the separation of vegetation types (e.g. woodland vs open woodland etc.) as tree density may have been an artefact of human activity (such as ring barking). Delineation of grassland from other vegetation types was initially based on canopy cover (i.e. grasslands had <5% canopy cover), but the tree cover threshold was varied in cases up to 10% cover based on the species in the herbaceous layer). ANOSIM and SIMPER tests in PRIMER version 6.1.8 (Clarke & Warwick 2006) on the cover data of the native, long-lived component of the vegetation at each site were used to test for significant differences in overall composition between vegetation types and to determine indicator species for each vegetation type. Cover data was log-transformed (log[x+1]) and a Bray–Curtis similarity matrix used for ANOSIM.

Ordination techniques were used to provide insights into the importance of environmental variables (soil attributes and rainfall) on native, long-lived perennial species composition and life-form dominance. Ordination of the native, long-lived perennial species cover data at each site was carried out in CANOCO 4.5 (ter Braak & Šmilauer 2002) and involved two steps: indirect gradient analysis (unconstrained ordination) and direct gradient analysis (constrained ordination). Indirect gradient analysis produced a gradient length of 4.5 (Table 2.1). A gradient length of this magnitude indicated linear methods such as principal components analysis (PCA) were inappropriate for analysis of the dataset as too many species deviated from the linear response to environmental gradients assumed by linear methods (Lepš & Šmilauer 2003).

The unimodal methods, detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA), were used. Indirect gradient analysis was used to investigate patterns in the floristic data regardless of independent (environmental) variables, and direct gradient analysis was used to explain the floristic data in relation to soil and rainfall data. The number of biplot axes for species–environment relations is restricted to two by default in DCA, hence the values for the third and fourth axes were not calculated or reported in Table 2.1. Species data were square-root-transformed and environmental variables were log-transformed (log[x+1]). The option to down-weight rare species was chosen.

Supplementary variables were computed from the floristic dataset and included cover and species richness of the dominant plant life-forms (trees [canopy species], shrubs [woody species not part of the canopy and not in the family chenopodiaceae], grasses, chenopods, forbs and graminoids [monocots not in the family Poaceae]). Vectors representing the dominant life-forms were projected onto a scatter plot of sites colour-coded by vegetation type. DCA highlighted differences in life-form dominance in the five vegetation types. A biplot of environmental and supplementary variables was produced from DCA to indicate environmental variables driving dominance by the various life-forms. An ordination plot of environmental variables was also obtained from CCA to identify gradients that distinguished the five vegetation types.

2.4 Results

Analysis of soil characteristics showed significant ($P < 0.05$) differences among vegetation types for pH, EC, Mg, K, EK_1 and EK_2, Na, silt and N (Table 2.1). River red gum sites were characterised by low pH, EC, EK_1 and EK_2, intermediate Mg, and high N and silt. Coolibah sites were characterised by high pH, Mg and Na, intermediate EC, EK_1 and EK_2, and low N and silt. Grassland sites were characterised by high pH and Na, intermediate Mg, EK_1 and EK_2 and silt, and low EC and N. Myall sites were characterised by high pH, Na, EC, EK$_1$ and EK$_2$, intermediate Mg and silt and low N. Black box sites were characterised by high pH and EC, intermediate Na, EK$_1$, EK$_2$ and silt and low Mg and N. No significant differences between vegetation types were found for ECEC, Ca, K, clay or sand.

Table 2.1: Mean values for soil variables for each vegetation type.

	River red gum (n = 13)	Coolibah (n = 17)	Grassland (n = 17)	Myall (n = 6)	Black box (n = 4)
pH^1	7.63b	8.74a	8.73a	8.88a	9.01a
EC (dS/m)^3	0.07c	0.20ab	0.21b*	0.28a	0.31a
Ca (cmol+/kg)	22.68a	21.02a	21.42a	18.95a	19.76a
Mg (cmol+/kg)	12.44ab	14.17a	12.62ab	12.16ab	9.74b
Na (cmol+/kg)^3	0.88b	2.85a	2.59a	3.15a	2.50ab
K (cmol+/kg)^1	0.92a	1.13a	0.91a	1.13a	1.48a
ECEC (cmol+/kg)	36.91a	39.17a	37.53a	35.38a	33.47a
EK$_1$	4.74c	5.56bc	6.96ab	8.65a	6.89abc
EK$_2$	2.58b*	2.85ab	2.36ab	3.18a	4.69ab*
N (%)^1	0.12a	0.07b	0.07b	0.08b	0.06b
Clay (%)^1	48.53a	54.92a	52.62a	50.24a	51.29a
Silt (%)^2	25.31a	17.18b	19.35ab	19.11ab	18.95ab
Sand (%)^2	26.16a	27.91a	28.03a	30.65a	29.77a

^{1}Rank transformed, ^{2}Log-transformed (Log(x+1) where applicable). ^{3}Square-root-ransformed to attain normality prior to AOV. $^\#$median values.*single outlier excluded. Letters denote significantly different groups using Tukey's multiple comparisons ($P \leq 0.05$).

A total of 127 native, long-lived perennial plant species were recorded during floristic surveys of the lower Namoi floodplain. Among the species recorded were six tree species, 17 shrubs, 37 grasses, 45 forbs, ten chenopods and seven graminoids (Appendix 1). ANOSIM confirmed a significant difference in composition of native, long-lived perennials between the five vegetation types (R = 0.56, P = 0.001). Pairwise tests indicated that the composition of most vegetation types differed significantly (P = 0.001) but grassland composition did not differ significantly from myall or black box sites (P > 0.05). SIMPER revealed characteristic species for each vegetation type (Table 2.2).

Table 2.2: Results of the SIMPER analysis of long-lived, perennial, native vegetation. Species are indicative of each vegetation type.

Vegetation type	Species	Av. % Abund	Av. Sim	Sim/SD	Contrib.%	Cum.%
River red gum (Average similarity: 29.76)	*Eucalyptus camaldulensis*	14.00	16.15	1.98	54.28	54.28
	Paspalidium jubiflorum	5.97	4.47	0.82	15.02	69.31
	Acacia stenophylla	3.85	1.56	0.38	5.26	74.56
	Carex inversa	1.44	1.55	0.55	5.21	79.77
	Paspalidium gracile	3.50	0.83	0.24	2.79	82.56
	Eucalyptus coolabah	2.92	0.69	0.27	2.33	84.90
	Boerhavia dominii	0.27	0.64	0.52	2.17	87.06
	Rumex brownii	1.21	0.47	0.30	1.58	88.64
	Acacia salicina	3.08	0.44	0.20	1.48	90.13
Coolibah (Average similarity: 39.61)	*Eucalyptus coolabah*	14.29	17.73	2.67	44.75	44.75
	Paspalidium jubiflorum	4.72	7.66	1.48	19.35	64.10
	Enteropogon acicularis	2.77	3.67	0.76	9.28	73.38
	Sporobolus mitchellii	0.75	1.60	0.66	4.04	77.42
	Acacia stenophylla	1.94	1.38	0.33	3.47	80.90
	Leptochloa divaricatissima	1.13	1.25	0.54	3.15	84.04
	Boerhavia dominii	0.32	0.68	0.47	1.73	85.77
	Alectryon oleofolius	0.53	0.65	0.35	1.65	87.42
	Goodenia fascicularis	0.46	0.65	0.52	1.63	89.06
	Asperula conferta	0.42	0.64	0.38	1.60	90.66
Grassland (Average similarity: 17.52)	*Panicum decompositum*	2.85	4.36	0.75	24.89	24.89
	Enteropogon acicularis	6.56	3.15	0.58	18.00	42.89
	Dichanthium sericeum subsp. *sericeum*	0.97	1.69	0.37	9.65	52.54
	Paspalidium jubiflorum	0.74	1.51	0.33	8.63	61.17
	Boerhavia dominii	0.30	1.48	0.54	8.48	69.65
	Astrebla lappacea	6.88	0.72	0.20	4.10	73.75
	Sida trichopoda	0.27	0.58	0.30	3.28	77.03
	Austrostipa aristiglumis	2.41	0.52	0.15	2.97	80.01
	Sporobolus caroli	0.47	0.45	0.21	2.56	82.56
	Eucalyptus coolabah	0.53	0.41	0.24	2.35	84.91
	Cyperus lucidus	0.27	0.40	0.23	2.28	87.20
	Paspalidium gracile	0.50	0.32	0.21	1.85	89.05
	Panicum queenslandicum var. *queenslandicum*	3.56	0.25	0.14	1.45	90.50
Myall (Average similarity: 41.48)	*Acacia pendula*	17.17	15.87	4.74	38.25	38.25
	Enteropogon acicularis	2.83	8.78	3.33	21.16	59.41
	Atriplex semibaccata	0.75	4.05	1.30	9.77	69.19
	Paspalidium gracile	2.00	3.89	0.79	9.38	78.57
	Sclerolaena calcarata	1.33	2.51	0.78	6.04	84.61
	Sporobolus caroli	0.73	2.33	0.98	5.61	90.22
Black box (Average similarity: 52.64)	*Eucalyptus largiflorens*	10.50	13.09	12.71	24.86	24.86
	Sporobolus caroli	4.00	7.45	11.98	14.15	39.01
	Sclerolaena calcarata	1.88	5.67	2.04	10.76	49.77
	Paspalidium gracile	3.01	4.33	0.96	8.22	57.99
	Atriplex semibaccata	1.50	3.75	0.87	7.12	65.11
	Muehlenbeckia florulenta	1.00	3.07	0.90	5.83	70.94
	Sclerolaena bicornis var. *horrida*	2.63	2.97	0.73	5.65	76.59
	Eremophila bignoniiflora	3.00	2.66	0.91	5.05	81.64
	Sida trichopoda	1.38	2.48	0.82	4.72	86.35
	Boerhavia dominii	0.45	1.91	1.44	3.63	89.99

Ave. Abund = the average abundance (i.e. projective foliage cover) of the species across all sites in that vegetation type. Av. Sim = average contribution of the species to the group Bray–Curtis similarity. Sim/SD = ratio of the average contribution divided by the standard deviation across the within-group similarities. Contrib% = contribution to Bray–Curtis similarity within the group. Cum% = cumulative % contribution to Bray–Curtis similarity within groups. Cut off for low species contributions was 90%.

Multiple compositional gradients were revealed in the floristic dataset, the longest gradient (4.5 on the first axis) explaining 11.3% of total species variability (Table 2.3). All axes showed strong correlations with the environmental (soil and rainfall)) data, especially the first and third axes. More than 50% of the variability in the floristic data was explained by the environmental variables (Sum of all eigenvalues/Sum of all canonical eigenvalues).

Table 2.3: Results from DCA of the native, long-lived, perennial flora, and environmental (rainfall and soil) variables at 57 sites across the lower Namoi floodplain.

DCA axes	1	2	3	4	Total inertia
Eigenvalues	0.608	0.395	0.289	0.151	5.389
Lengths of gradient	4.543	3.842	2.988	2.794	
Species–environment correlations	0.933	0.795	0.896	0.696	
Cumulative percentage variance					
of species data	11.3	18.6	24.0	26.8	
of species–environment relations	19.0	27.9	–	–	
Sum of all eigenvalues					5.389
Sum of all canonical eigenvalues					2.774

The DCA ordination showed that sites tended to cluster by vegetation type as a result of differences in the composition of the native, long-lived, perennial component of the vegetation (Figure 2.9). Differences in life-form among sites and vegetation types were also evident. River red gum sites had greatest graminoid species richness and cover, a high cover of trees, forbs and shrubs, and low chenopod richness and cover. Coolibah woodlands showed high levels of tree cover and richness, low chenopod cover and richness, and low shrub richness. The understorey in these communities was dominated by grass cover, but forb richness was also high. Myall shrublands and black box open woodlands showed high chenopod richness and cover. Grasslands were characterised by high grass cover and richness, and high forb richness.

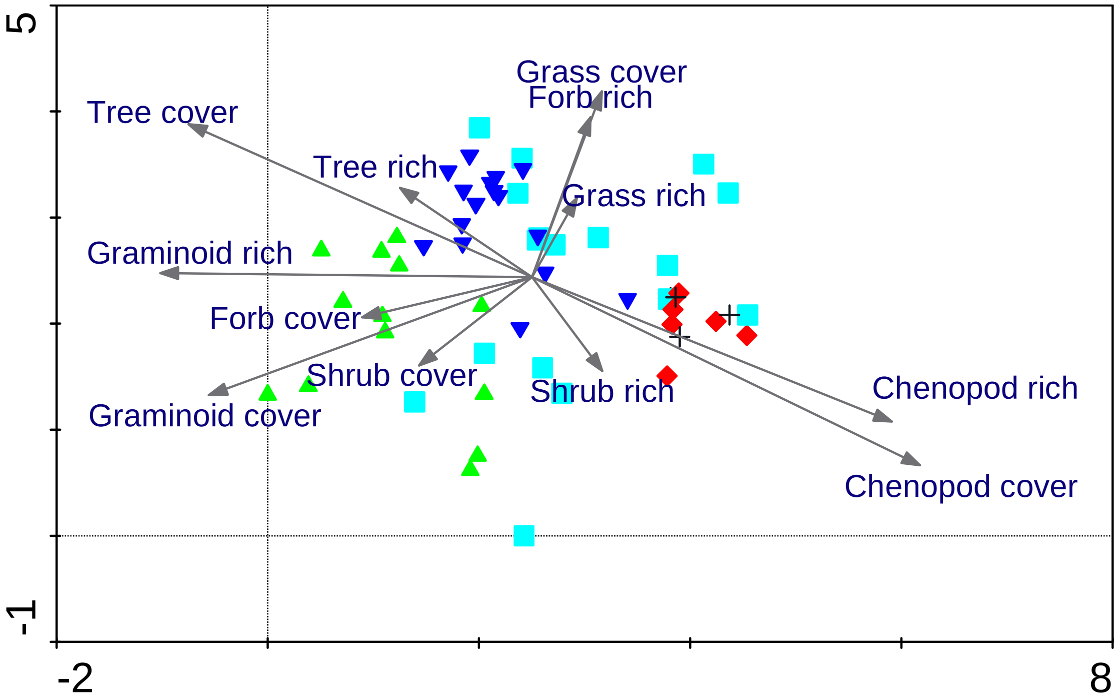


Figure 2.9: DCA ordination illustrating differences in dominant life-forms of native, long-lived, perennial vegetation for five vegetation types on the lower Namoi floodplain. ▲ = river red gum sites, ▼ = coolibah, ♦ = myall, '+' = black box and ■ = grassland; rich = richness.

DCA ordination of life-form data and environmental variables showed a number of links between the two (Figure 2.10). Chenopod species richness and cover were higher at sites with higher soil EC, EK_1, EK_2 and sand percentage; these sites also showed low levels of N. Grass species richness and cover, and forb richness vectors clustered with soil Na and clay percentage. Forb cover was highest in high rainfall sites with high soil Ca and N, and low values of pH, EC, Na and EK_2, as was graminoid cover and richness, and shrub cover. Both tree species richness and cover fell between the Ca and Mg vectors.

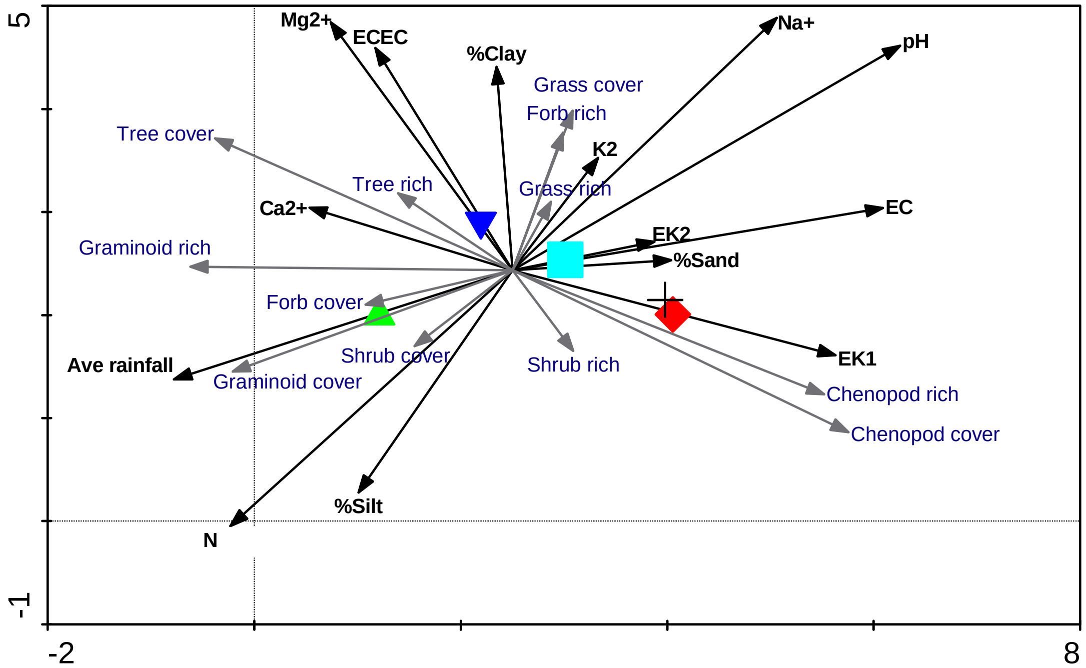


Figure 2.10: DCA ordination of the relationship between environmental variables and different plant life forms in the native, long-lived, perennial component of the vegetation on the lower Namoi floodplain. The five coloured symbols represent the centroid of each of the five vegetation types. ▲ = river red gum sites, ▼ = coolibah, ♦ = myall, '+' = black box and ■ = grassland; rich = richness.

CCA showed very clear species–environment correlations, with all four axes showing values over 0.9 (Table 2.4). The first environmental axis was the most influential, explaining 20% of the variance. The percent variance in both the species and species–environment data was similar in the DCA and CCA, indicating that the environmental variables measured were the most important in determining floristic composition.

Table 2.4: Results from CCA for native, long-lived, perennial flora, and environmental (rainfall and soil) variables at 57 sites across the lower Namoi floodplain.

CCA axes	1	2	3	4	Total inertia
Eigenvalues	0.551	0.356	0.309	0.267	5.389
Species–environment correlations	0.962	0.937	0.917	0.905	
Cumulative percentage variance					
of species data	10.2	16.8	22.6	27.5	
of species–environment relations	19.9	32.7	43.8	53.4	
Sum of all eigenvalues					5.389
Sum of all canonical eigenvalues					2.774

The CCA (Figure 2.11) and DCA (Figure 2.9) ordinations were similar, the placement of environmental axes and sites being almost identical. The CCA ordination (Figure 2.11) indicated that river red gum sites occurred on siltier soils with low pH and Na, and high N. Black box and myall preferred sandier soils with high EC and K, and low ECEC, Mg and Ca, most sites occurring at the low-rainfall (i.e. western) end of the catchment. Coolibah and grassland soils had a higher clay content and ECEC, which is likely correlated with higher concentrations of divalent cations (Ca and Mg), and a lower EC and pH than soils associated with myall and black box communities.

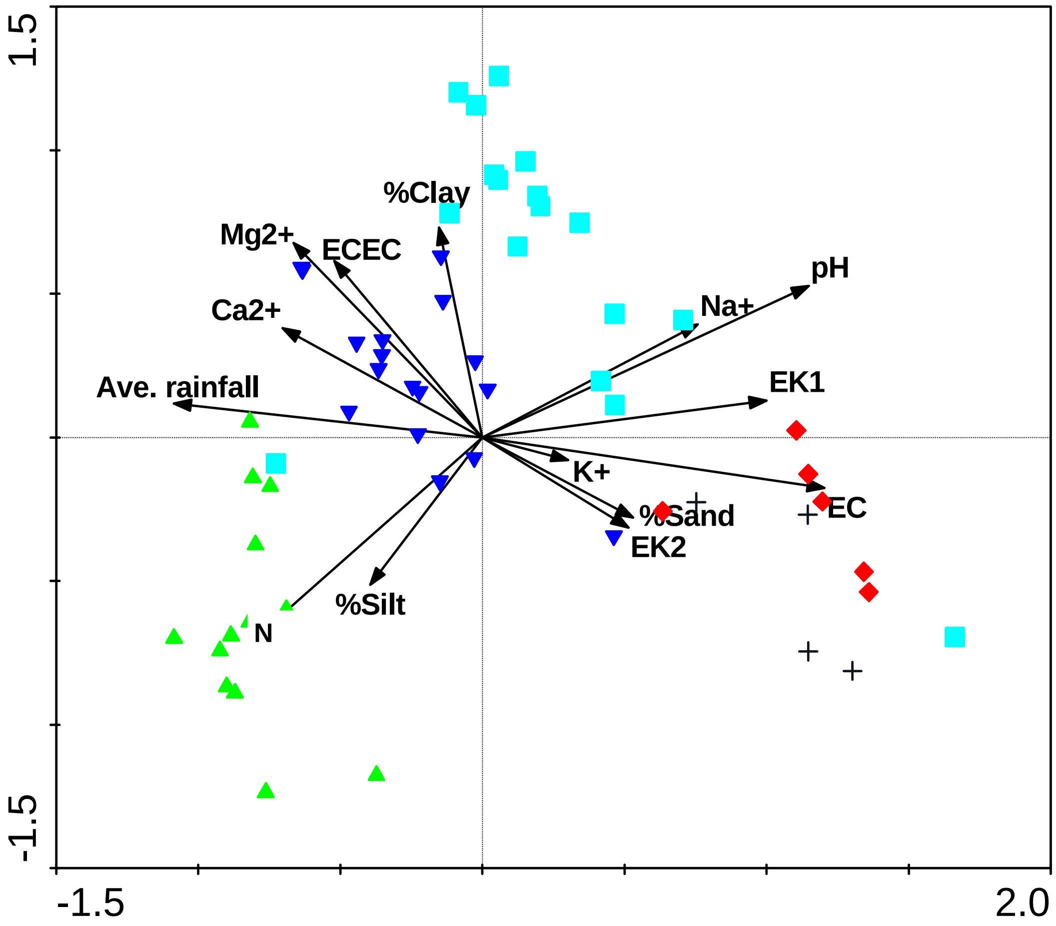


Figure 2.11: CCA ordination showing the environmental variables driving long-lived, native, perennial vegetation patterns across the lower Namoi floodplain. ▲ = river red gum sites, ▼ = coolibah, ♦ = myall, '+' = black box and ■ = grassland.

2.5 Discussion

2.5.1 Vegetation of the Lower Namoi Floodplain

The five vegetation types in this study showed little overlap in either the DCA or CCA ordinations, indicating that they are largely distinctive in terms of native, perennial species composition. The percentage variance of the species–environment relationship represented the variability in the floristic data explained by the environmental data. Environmental variables accounted for more than 50% of the variability in species composition, a large proportion given the number of species in the dataset. Strong correlations on multiple axes in the CCA suggested that at least four environmental gradients governed species composition. A floristic description of each vegetation type and the dominant environmental drivers dictating distribution across the study region follows. From this information, preferred landscape positions for each vegetation type are inferred.

2.5.1.1 River Red Gum

The river red gum group of sites was characterised by dominance of river red gum in the canopy, occasionally with coolibah. The understorey was variable; at some sites, a well developed shrub layer consisting of *Acacia* species such as river cooba (*Acacia stenophylla*) and cooba (*A. salicina*) was present and the herbaceous vegetation was dominated by forbs such as tarvine (*Boerhavia dominii*). At other sites, the understorey consisted of a grassy layer largely dominated by Warrego summer grass (*Paspalidium jubiflorum*) and slender panic (*Paspalidium gracile*). At most sites, a large proportion of the understorey was made up of species of Cyperaceae (e.g. *Carex* or *Cyperus*) and swamp dock (*Rumex brownii*). The species composition of this group of sites fits within

Benson's (2006) 'River red gum open forest and woodland mainly of the Darling Riverine Plains Bioregion' (ID36). The present study concentrated on native, long-lived, perennial species to reduce the effects of management on the results, but it appears that a feature of this community may be the presence of a large number of annual and short-lived perennial species in the understorey (Chapter 6).

The soils of the river red gum sites were siltier and contained more N and Ca, and had lower Na content, pH and EC. While not included in the analysis due to collinearity with N, total organic carbon (TOC) was also high in river red gum sites. Although the ordinations suggested that annual rainfall influences the distribution of river red gum sites, the community is spread along the entire length of the Namoi River and the result was an artefact of the lack of sites at the western end of the catchment, and the greater proportion of sites south-east of Narrabri in this group than in other vegetation types.

River red gum sites were mostly confined to the banks of the Namoi River. Exceptions to this were the depressed floodplain between Narrabri and Wee Waa characterised by shallow flood runners (shallow channels that conduct water in times of flood), and two sites east of Narrabri that were meanders of the Namoi River. Debris is more likely to be deposited by floods at these sites than any other floristic group, accounting for both the siltier soils and the higher levels of N and C in the soil. Riparian areas are naturally more productive in terms of terrestrial vegetation than floodplain sites away from the river due to the greater permanence of surface water and, in some cases, ground water supply. Higher N at some sites was probably also due to the presence of the nitrogen-fixing, *Acacia*-dominated, shrub layer. High TOC and Ca were probably due to the semi-deciduous nature of river red gum trees and their tendency to drop large amounts of leaf litter during periods of stress (DSE 2006). Many of the river red gums were very large trees, some >3 m in DBH, and these trees regularly shed large branches and woody debris. Due to a high C:N ratio, wood is broken down slowly compared to foliage and herbaceous plant matter, and remains as TOC in the soil for many years. Some of the river red gums were of such a size that it is likely that they have been influencing soil properties for centuries. Organic matter is known to increase soil ECEC (McKenzie et al. 2004). This probably accounts for the elevated levels of Ca in river red gum soils.

2.5.1.2 Coolibah

The coolibah group was characterised by a canopy dominated by coolibah trees with scattered rosewood (*Alectryon oleofolius*) and river cooba tall shrubs. The herbaceous layer often showed a well developed grass component in many cases, the most common grasses being Warrego summer grass, curly windmill grass (*Enteropogon acicularis*), rat's tail couch (*Sporobolus mitchellii*) and *Leptochloa divaricatissima.* Forb species richness was also high in this community as indicated by the DCA ordination, tarvine, common woodruff (*Asperula conferta*) and *Goodenia fascicularis* in particular were identified as important by SIMPER. The coolibah group was best aligned with Benson's (2006) 'Coolabah – River Coobah – Lignum woodland of frequently flooded channels mainly of the Darling Riverine Plains Bioregion' (ID39), but most sites were not in frequently flooded channels, but on the floodplain. Three outlier sites in the DCA ordination, with higher chenopod cover than the majority of coolibah sites, were more consistent with Benson's (2006) 'Coolabah open woodland with chenopod/grassy ground cover on grey and brown clay floodplains.'

The coolibah group occurred on soils with the highest clay and Mg content, and ECEC and relatively high Ca levels. Soil pH of the predominantly grey or brown clays in coolibah sites was alkaline with a mean of 8.74 and calcium carbonate nodules were usually present. The soil often showed subtle gilgai formation, usually crabhole gilgai, and this microtopography contributed to increased plant species richness with different plant assemblages in the bottom of the gilgai compared to the top. This is most likely as a result of differences in available soil moisture between the top and bottom of the gilgai (Beadle 1981). The high clay content, which was often dominated by montmorillinite (R. Smith unpub. data), contributed to a strong self-mulching behaviour in these soils, with cracks up to 1 m or more deep in dry soil. This action of the soil probably has a large effect on the structure of the community by breaking off the roots of small trees and shrubs (Beadle 1981; Lang 2008), thereby encouraging an open structure.

Landholder observations indicated that coolibah sites are regularly flooded and regeneration is observed after prolonged floods, though it appears that the last major widespread recruitment was some time ago, perhaps as long ago as the 1970s (Maher 1995). Evidence of regular flooding was provided by the vegetation, species such as rat's tail couch, Warrego summer grass, *Leptochloa divaricatissima,* river cooba and lignum preferring wetter, often gilgaied floodplain soils (Cunningham et al. 1992; Jacobs et al. 2008). The canopy of this community tended to be denser in drainage lines and areas liable to inundation, though the species richness of the canopy and shrub layers was greater in less flood-prone areas. Coolibah-dominated vegetation is listed as an endangered ecological community 'Coolibah–Black Box Woodland of the northern riverine plains in the Darling Riverine Plains and Brigalow Belt South bioregions' under the NSW *Threatened Species Conservation Act 1995* (NSW Scientific Committee 2004). 'Coolibah (*Eucalyptus Coolibah*) / Black Box (*E. largiflorens*) woodlands of the Northern NSW Wheatbelt and Queensland Brigalow Belt bioregions' is currently being considered for listing under the *EPBC Act 1999* (DEWHA 2010).

2.5.1.3 Myall

The myall group was characterised by a canopy of myall shrubs and a mixed grass and chenopod herbaceous layer dominated by curly windmill grass, slender panic, fairy grass (*Sporobolus caroli*), berry saltbush (*Atriplex semibaccata*) and redburr (*Sclerolaena calcarata*). A defining characteristic of this community, aside from the dominance of myall in the canopy, was the species richness and cover of chenopods in the herbaceous layer. The myall group defined in this study fits within Benson's (2006) 'Weeping Myall open woodland of the Darling Riverine Plains and Brigalow Belt South Bioregions' (ID 27).

The soil characteristics driving the distribution of myall in the study region included high sand content, EC and exchangeable K, and low Ca, Mg and ECEC. The soils of myall sites were broadly brown clays. X-ray diffraction (XRD) analysis suggested that, unlike the grey clays of the region, brown clay soils lack the clay mineral, montmorillinite, and may instead be dominated by illite (R. Smith, unpub. data). The chemical analysis provided further evidence to support this with a low ECEC and high K, characteristics of an illite-dominated soil (Gourley 1999). Surface characteristics also provided evidence to suggest these soils were low in montmorillinite. The soil surface was not self-mulching; rather, it was often crusted and had a massive fabric. The abundance of Na in these soils also needs to be taken into account when interpreting the behaviour of the surface soil, as sodic soil is more prone to dispersion.

Landholder observations indicated that the myall sites are rarely flooded, if at all, and this was reflected in the soil characteristics. The 'sandier' nature of the myall sites (bearing in mind that the clay content at 20–30 cm deep was still over 50%) indicates that the slow-moving floodwaters that cover much of the rest of the floodplain in times of flood and deposit fine sediments have not done so at myall sites. Evidence has also been reported by authors such as McGarity (1950) that suggest these soils are older than the grey clays. This implies that myall occurs on slightly elevated parts of the floodplain in the study region, as opposed to its preferred position on the Liverpool Plains along drainage lines and in swamps (Banks 1995). While myall was only recorded on elevated brown clay rises in this study, perhaps it once occurred in lower positions on the grey clays as indicated by Benson (2006), but has been cleared. Extensive clearing has resulted in this community being listed as an endangered ecological community under both the NSW *Threatened Species Conservation Act 1995* 'Myall Woodland in the Darling Riverine Plains, Brigalow Belt South, Cobar Peneplain, Murray-Darling Depression, Riverina and NSW South western Slopes bioregions', and the federal *Environmental Protection and Biodiversity Conservation Act 1999* 'Weeping Myall Woodlands'.

2.5.1.4 Black Box

This group was characterised by a canopy dominated by black box, occasionally with a tall shrub layer dominated by eurah (*Eremophila bignoniiflora*). The herbaceous understorey was dominated by chenopods, in particular goat-head burr (*Sclerolaena bicornis* var. *horrida*), redburr and berry saltbush, but grasses such as fairy grass and slender panic and small, sparse lignum plants were also present. Pin sida (*Sida trichopoda*), tarvine and clumped bindweed (*Polymeria longifolia*) were among the more abundant forbs recorded. The defining characteristic of this community, aside from the dominance of black box in the canopy, was the abundance of chenopods in the understorey, many of which were not included in this chapter because they were annual or biennial species. The average similarity for this group was high due to the small number of sites and tight distribution at the western-most, low-rainfall end of the study region. The black box group reported here fits within Benson's (2006) 'Black Box woodland on floodplains mainly in the Darling Riverine Plains Bioregion' (ID 37).

The soils of the black box group were similar to that of the myall group. The soils had high pH, EC and K, and were 'sandy.' The soils were low in Mg, ECEC and N compared to soils of the other communities. Like the myall sites, the clays of the black box group appeared to be illite-dominated (R. Smith, unpub. data) and this was reflected in the soil chemistry. This suggests that the black box sites are less prone to inundation.

2.5.1.5 Grasslands

Grasslands were characterised by low canopy cover of trees, only the occasional coolibah tree being present, and high grass cover in good seasons. The dominant grass species were native millet (*Panicum decompositum*), curly windmill grass, Queensland bluegrass (*Dichanthium sericeum* subsp. *sericeum*), Warrego summer grass, curly Mitchell grass (*Astrebla lappacea*), plains grass (*Austrostipa aristiglumis*), fairy grass, slender panic and coolibah grass (*Panicum queenslandicum* var. *queenslandicum*). Forb species richness was high, with tarvine and pin sida being the most important (SIMPER). A graminoid (*Cyperus lucidus*) also contributed to group similarity. The average similarity was low (17.5) for this community, and it is likely that this community should be divided into several distinct communities. Two outliers in the grasslands group appeared in the CCA ordination. A grassland that occurred with the myall–black box group was highly sodic (ESP = 21.04 at 20–30 cm depth) and the majority of grass species recorded during floristic surveys were annual or short-lived perennials. The other outlier occurred with the river red gum group: this site was adjacent to a stand of river red gum trees, was low in the landscape and had high N and silt percentage values.

A DCA biplot of the common grass species and environmental variables recorded in this study (Figure 2.12) suggested that each species occupies a different environmental niche. Even species that commonly co-occur such as Mitchell grass and Queensland bluegrass occurred on opposite sides of the ordination, Mitchell grass preferring soils with higher pH, EC and Na, and Queensland bluegrass preferring lower pH and Ca-rich soil. Perhaps these species exist together because they occupy different microhabitats created by the gilgai phenomenon commonly occurring in these cracking clay soils, higher pH being common on the mound of the gilgai (Stace et al. 1968).

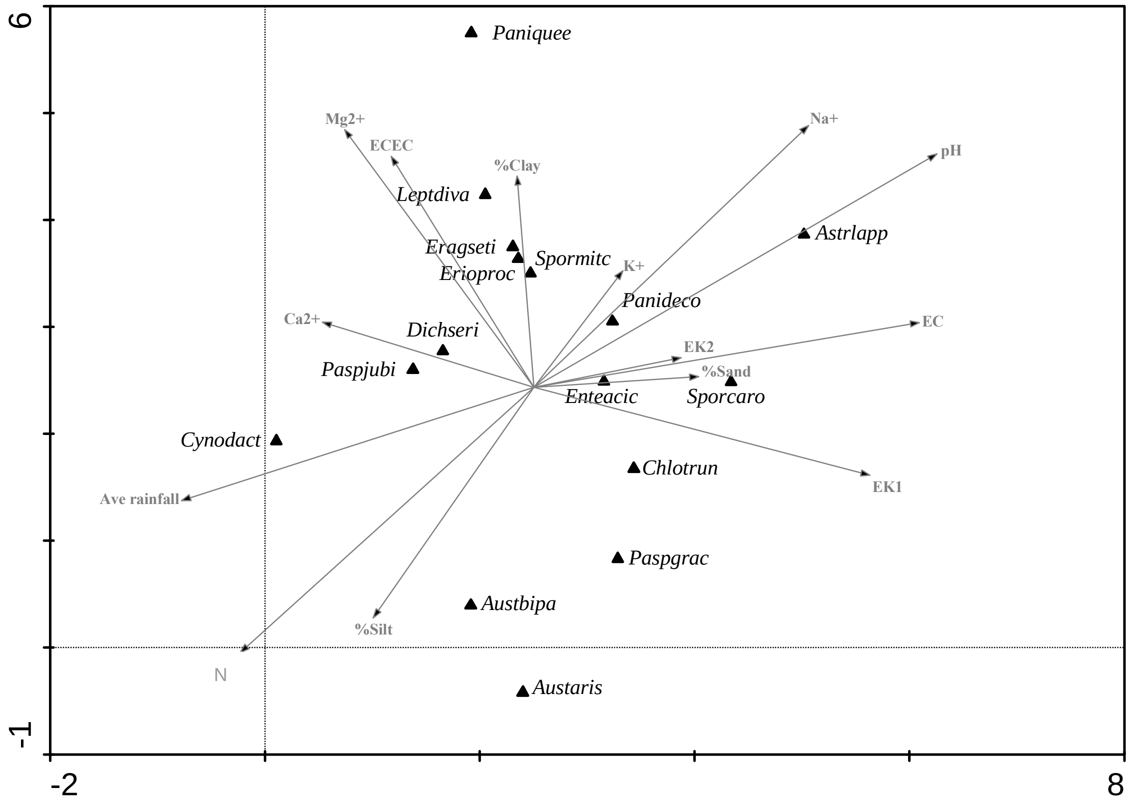


Figure 2.12: DCA biplot showing distribution of common grass species in relation to soil and rainfall gradients. The first four letters of the genus name and specific epithet make up the codes used in the figure for the following species: *Astrebla lappacea, Austrodanthonia bipartita, Austrostipa aristiglumis, Chloris truncata, Cynodon dactylon, Dichanthium sericeum* subsp. *sericeum, Enteropogon acicularis, Eragrostis setifolia, Eriochloa procera, Leptochloa divaricatissima, Panicum decompositum, Panicum queenslandicum* var. *queenslandicum, Paspalidium gracile, Paspalidium jubiflorum, Sporobolus caroli* and *Sporobolus mitchellii.*

The most frequent species were located towards the centre of the ordination indicating that their occurrence is not driven by any particular environmental variable. Separation of the grasslands dominated by curly Mitchell grass, plains grass and coolibah grass might be a good place to start in recognising individual communities. Benson (2006) recognised at least three distinct communities that fit the species composition of the grasslands sampled in this study. The ‘Mitchell Grass grassland of the semi-arid (hot) and arid zone alluvial floodplains’ (ID43), ‘Plains Grass grassland on alluvial dark grey clays of central New South Wales’ (ID45), and the ‘Queensland Bluegrass – Cup Grass – Mitchell Grass – Native Millet alluvial plains grassland’ (ID 52) were represented, although many of the species listed in Benson's communities were not recorded in this study. An additional community dominated by Warrego summer grass, *Leptochloa divaricatissima,* slender rat's tail grass (*Sporobolus creber*) and native millet, probably derived by clearing of coolibah woodland on heavy grey clays, and a coolibah grass-dominated community also on heavy grey clays, warrant consideration as additions to Benson's classification.

2.5.2 Further Research

While a large amount of the variation in the species composition data was explained by the environmental variables, a large amount remained unexplained. One aspect that was not investigated in this study and probably accounts for some of the unexplained variability is water relations. No attempt was made here to address the possible influence of flood regimes (frequency, duration etc.) or groundwater availability. Being a floodplain in a semi-arid region, it is likely that water plays a major role in dictating the distribution of plant communities across the landscape. Floods occur on a regular basis on the lower Namoi floodplain although the distribution and movement of floodwater across the plain is currently poorly documented. Studies of flood movement patterns across the floodplain, especially around Wee Waa, have been carried out (e.g. Burton et al. 1977), and many pre-date the extensive development for irrigated agriculture that has taken place in recent years. However, detailed mapping of floodwater distribution and movement was not available. Accounts of floodwater movement were obtained from landholders, but in many cases were not detailed or reliable enough to quantify inundation frequency at our sites. Current flood patterns are likely to have changed through time due to changes in land use and flood behaviour.

This further complicates data interpretation as conditions that were in play during the establishment of some of the vegetation may not now occur. At a finer scale, runoff and runon patterns may have changed due to disturbances such as grazing, tree removal and road construction.

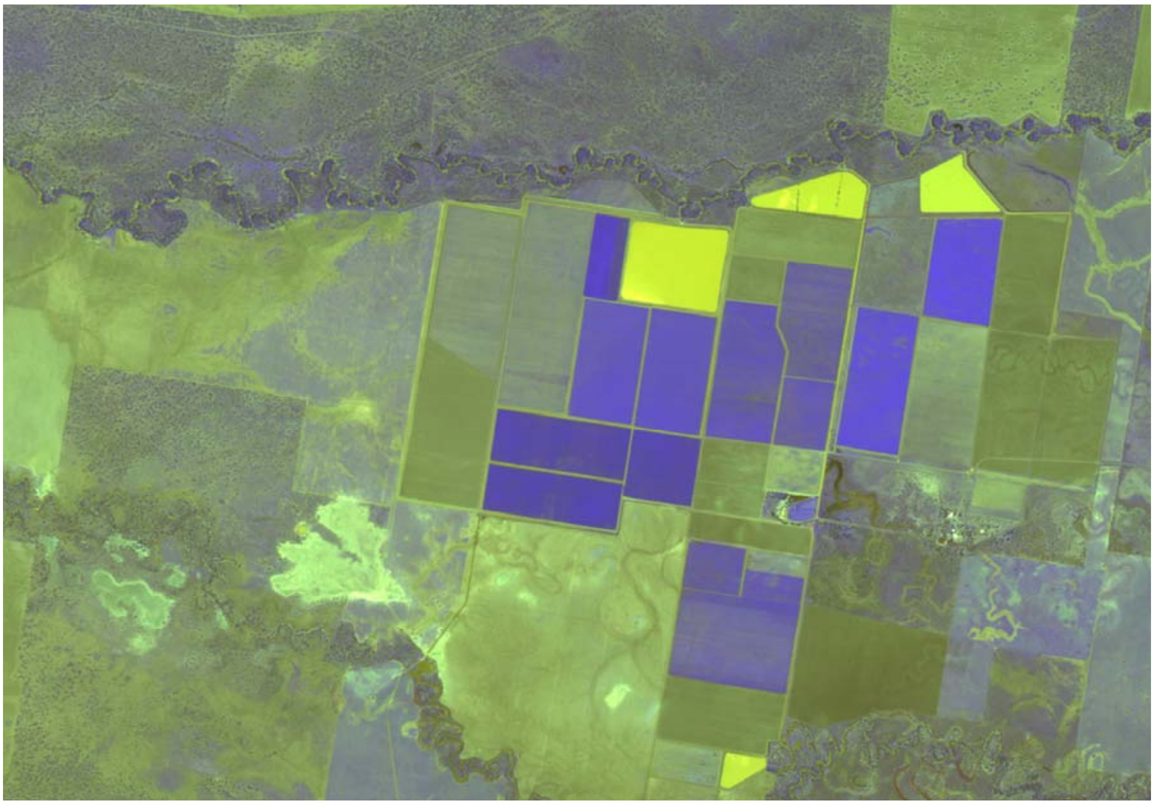
Detailed groundwater mapping has not been carried out for large areas of the lower Namoi, although it is present as many landholders use groundwater for irrigation or stock water. Provision of permanent water for stock in the form of bores and wells in the area has allowed for greater grazing pressure in areas far from surface flowing creeks and rivers, and allowed stock to remain in areas during drought conditions. A cap and pipe program of unused bores and bore drains was recently introduced in the area to make stock watering more efficient (DNR 2006). There has been little documentation of the effects of grazing and the associated construction of bores and drains on the vegetation of the study region. The cap and pipe program provides an opportunity to document vegetation recovery.

Further evidence of the presence and potential importance of groundwater in the region is given by the size of some of the older eucalypt trees in surface-water limited situations. Coolibah trees, in particular, may access water from groundwater aquifers. Average rainfall figures alone across much of the distribution of coolibah are too low to allow trees to attain the dimensions that many do (Beadle 1948). If coolibah trees rely purely on rainfall and floods, climate change and increased demand for water for agriculture in the study region may have a debilitating effect on this community. Climate change may also alter weather patterns to the extent that the conditions required for reproduction and persistence of these communities no longer occur. The episodic nature of recruitment in many long-lived species such as coolibah and river red gum suggests that specific regeneration requirements, most likely related to flooding, must be met for recruitment to occur. Studies of the dependence of floodplain vegetation on various water sources are a priority, as the longevity and persistence of these communities is at stake.

2.6 Conclusions

The dominant vegetation types of the lower Namoi floodplain occupy different environmental niches at a range of scales. Differences in soil properties, flood frequency and distribution, and annual rainfall create conditions that differentially favour river red gum, coolibah, myall, black box and grassland communities, and influence the dominance of different life-forms in each community. The key environmental factor separating each of the communities is the differential availability of water in space and time across the landscape. Soil characteristics have an overriding influence on water and nutrient availability and the vegetation communities vary with a wide range of soil characteristics. Groundwater and flood patterns are probably important, which could be problematic due to the value of water in agriculture-dominated landscapes and the predicted impacts of climate change.

Chapter 3: Single-date SPOT 5 Imagery Accurately Maps Woody Vegetation Extent on a Semi-arid Floodplain



SPOT 5 false colour image of a cotton farm north-west of Pilliga (image captured 21 January 2005).

Chapter 3: Single-date SPOT 5 Imagery Accurately Maps Woody Vegetation Extent on a Semi-arid Floodplain

3.1 Introduction

The extent and distribution of woody vegetation has important implications for many facets of natural resource management (Cihlar 2000). The Convention on Biological Diversity (Strand et al. 2007), Kyoto Protocol (United Nations 1998) and the Montreal Process (Anon. 2007) and their implementation by signatory states all require thorough knowledge of landcover and vegetation resources. At a local scale, vegetation maps showing the distribution of vegetation types inform land managers and administrators about biotic and abiotic land attributes, including fauna, soil type, flood and other disturbance regimes (Beadle 1981), land use potential and value. In over-cleared landscapes, vegetation maps provide data for restoration planning such as priorities for revegetation to increase connectivity of existing landscape elements (Apan et al. 2002). As vegetation is a key provider of many ecosystem services (Kremen 2005), knowledge of the spatial and temporal distribution of vegetation across landscapes, catchments and bioregions is important for land and resource use planning (Kremen et al. 2007).

Aerial photograph interpretation (API) and extensive ground survey were traditionally employed in the preparation of vegetation maps (Zonneveld 1988). Manual classification of satellite imagery was later introduced and in some cases replaced API (e.g. Woodgate & Black 1988). However, while both these approaches can produce high quality maps, they are slow, labour-intensive and expensive (Belward et al. 1990; Bird et al. 2000). For vegetation mapping over larger areas (e.g. large catchments of the order of 10 000 km^2, states, whole countries or the globe), the excessive cost of traditional approaches necessitates the use of precise and accurate semi-automated remote sensing methods (Running et al. 1994; Cihlar 2000; Cox et al. 2001). Satellite data permit semi-automated production of vegetation maps, thereby extending the area that can be mapped in a short space of time (Bird et al. 2000). Several generations of satellites have been launched that can be used to map vegetation landcover. The Landsat and SPOT satellites provide a long-term, continuous series of data and have medium-resolution sensors designed specifically to monitor land (Jensen 2005).

Unsupervised and supervised classifications of single-date satellite imagery are widely used to map landcover (Cihlar 2000). Unsupervised classification methods are generally more accepted in mapping large areas as little prior knowledge of the landcover types in the area is required initially. In contrast, supervised techniques require detailed knowledge of the area being mapped before the procedure can commence, as user-defined training classes representing each landcover class form the basis of the classification. Single-date imagery limits the accuracy of final mapping products as season and the response of vegetation to weather conditions leading up to image capture influence the spectral characteristics of the vegetation. The ever-changing appearance of agricultural landscapes imposes further limitations, as landcover changes can be sudden and drastic as a result of crop harvesting, planting or cultivation. The NSW government has recently obtained single-date SPOT 5 imagery for much of NSW and this data was made available as an in kind contribution for this study. Additional imagery was available but prohibitively expensive.

Numerous projects have used Landsat or SPOT data to map and monitor vegetation resources over large areas using a range of approaches with varying success. Barson et al. (2000) combined state agency publications to document changes in landcover across Australia between 1990 and 1995. Unsupervised classification with 100–150 classes was used by most of the state agencies to classify Landsat TM data and map woody vegetation. The final accuracy achieved by each state agency using this approach was ≥90%. This study provided a landcover map for approximately 40% of the continent, with a focus on agricultural areas where landcover change occurs rapidly. While the mapping projects incorporated by Barson et al. (2000) achieved a high degree of accuracy, no justification was provided to explain why more than 100 unsupervised classes were required to map woody vegetation. The number of classes used in an unsupervised classification has a direct effect on the cost of mapping projects in terms of computer time and power, and labour. However, very few studies justify the choice of method used in terms of cost-effectiveness (Green et al. 1996). The question arises, is a similar level of accuracy achievable with less than 100 classes? Is it possible to make large-area projects more time and cost-efficient?

This project investigated the use of unsupervised classification to map woody vegetation from single-date SPOT 5 imagery across a semi-arid floodplain in north-western NSW in order to locate woody vegetation communities for further fieldwork in the absence of more accurate data. Up-to-date high resolution vegetation maps that show the location of remnants, illustrate their landscape context and thereby provide information on their condition, were considered necessary for farm and catchment planning and management. Three questions were addressed: (1) how accurately can unsupervised classification of SPOT 5 data map tree species in a small test site, (2) what is the optimal number of unsupervised classes for mapping woody vs non-woody landcover, and (3) what level of accuracy can be attained using unsupervised classification of SPOT 5 data to map woody vs non-woody vegetation across a large region (ten SPOT 5 scenes). A number of limitations were imposed on this study, such as a necessity for the project to be completed within a short timeframe with few resources and use of single-date imagery. These limitations are discussed in the light of the findings of the study.

3.2 Methods

3.2.1 Study Region

The study was undertaken on the floodplain of the Namoi Valley, in northern NSW (Figure 3.1). This area was chosen as there is a lack of high resolution vegetation maps covering the area, only small areas have been mapped at a resolution useful for farm-level land use planning, and most maps pre-date current vegetation legislation banning broad-scale land clearing. The Namoi floodplain, as defined by the area inundated in the largest recorded flood, has been extensively developed and cleared for agriculture, both cropping (especially cotton and wheat) and grazing (beef cattle and sheep). Steep basalt hills and areas of low agricultural value constitute most of the remaining woody vegetation in the region (Mawhinney 2004), most of which is under pressure from surrounding intensively managed landscapes.

Rainfall across the lower Namoi floodplain decreases from approximately 600 mm near Boggabri in the east to 400 mm near Walgett in the west, with a slight summer dominance (Bureau of Meteorology 2007). Rainfall is unreliable, often falling sporadically as thunderstorms or as a result of depressions originating from tropical cyclones that cross the southern Queensland coast (Kuhnell et al. 1998; SPCC 1980). A flood resulting from unusually high rainfall was experienced across the study region in December 2004 (Bureau of Meteorology 2007), during the time of image capture. This rainfall event resulted in a flush of growth of summer-active grasses and standing water over large areas of low-lying land for several weeks into January 2005.

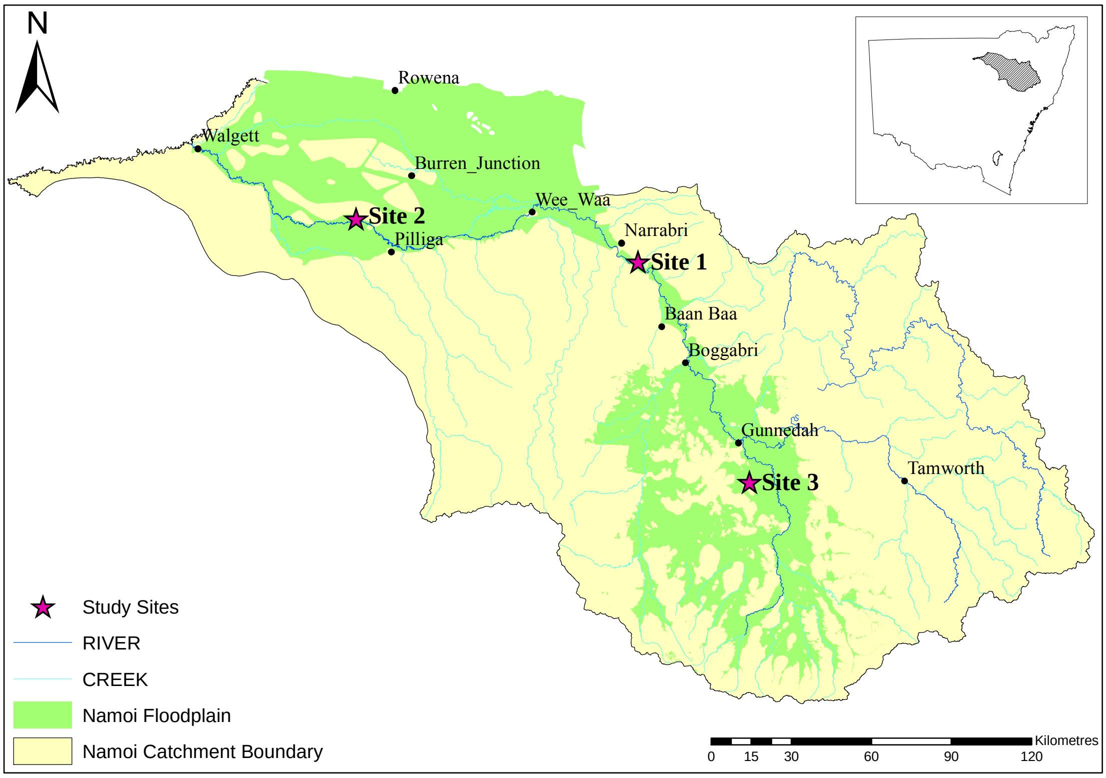


Figure 3.1: Namoi catchment in northern NSW, indicating the extent of the floodplain study region (defined by the largest recorded flood) and study sites.

The floodplain is dominated by grey to brown, fertile, cracking clay soils. Areas of coarser textured soils, referred to as ‘red ridges’ (prior stream formations), dissect the floodplain, are rarely flooded and support a flora different to the rest of the floodplain. These areas are rarely cleared due to their lower fertility and greater erosion potential compared to the clay soils of the rest of the floodplain. Dominant woody vegetation communities in the study region are outlined in Table 3.1.

Table 3.1: Dominant woody vegetation types of the mid–lower Namoi floodplain.

Dominant woody species	Landscape position	Structure	Associated species
River red gum (*Eucalyptus camaldulensis*)	Along rivers, major drainage lines, swamps and billabongs	Open grassy or shrubby forest, grassy or shrubby woodland, open grassy woodland	*Acacia stenophylla,, A. salicina, Muehlenbeckia florulenta* (shrubby riparian community), annual and perennial grasses (grassy woodland community)
Coolibah (*Eucalyptus coolabah* subsp. *coolabah*)	Minor watercourses, low-lying areas subject to regular flooding	Dense woodland, savanna-like grassy woodland	*E. camaldulensis, E. largiflorens*, perennial grasses and chenopod spp.
Myall (*Acacia pendula*)	Shallow depressions far from active drainage channels	Isolated trees, grassy – open grassy woodland	*C. cristata*, *E. populnea*, *E. largiflorens*, perennial grasses and chenopod spp.
Poplar box (*Eucalyptus populnea*)	Higher floodplains, minor flattish watercourses	Grassy woodland, open grassy woodland	Usually almost pure stands on floodplain soils with perennial grasses, with *Geijera parviflora* Lindl. or *C. cristata*
Yellow box/Blakely's red gum/white box/grey box (*E. melliodora*/*E. blakelyi/E. albens./E. moluccana*)	Floodplains and elevated stony areas (species dominance changes with landscape position	Grassy woodland, grassy open woodland	*G. parviflora*, annual and perennial grasses
Black box (*Eucalyptus largiflorens*)	Secondary or lesser flooded floodplains, usually on brown clay soils in the west of the study region	savanna-like woodland with chenopod-dominated understorey	*E. coolabah, Eremophila bignoniiflora, Sclerolaena* spp.

3.2.2 Satellite Data

SPOT 5 multispectral data in five bands (B1: green, 0.50–0.59 µm; B2: red, 0.61–0.68 µm; B3: near infrared, 0.78–0.89 µm; B4: mid infrared (MIR), 1.58–1.75 µm, 10 m pixels) (Satellite Imaging Corporation 2003) was used in mapping, and pan sharpened colour composite imagery (2.5-m pixels) was used in classification. Imagery was obtained under licence from the Department of Natural Resources (DNR). Radiometric correction of differences in sensitivity of the elementary detectors of the viewing instrument had been carried out (M. Shaikh, pers. comm., 8 August 2006). The image had also been geometrically corrected using a resampling model that compensated for systematic distortion effects and performed the necessary transformations to tie it to a geographic reference (SPOT Image 2003). The data were supplied in GDA94 zones 55 and 56. Orthorectification using a digital elevation model (DEM) was also conducted to correct for residual parallax errors due to relief displacement (SPOT Image 2003). As a result of these corrections, the locational error of the imagery was <15 m.

Images consecutively covering a 60 × 60 km grid across the study region were captured on cloud-free passes of the SPOT 5 satellite between November 2004 and May 2005 (a single 60 km × 60 km image was captured for each section of the region) (Table 3.2). The classification of woody vegetation communities was conducted on a subset of the SPOT 5 imagery near Narrabri (Figure 3.2). Two additional image subsets at Pilliga (Figure 3.3) and Curlewis (Figure 3.4) were used to determine the optimal number of unsupervised classes required to map woody versus non-woody vegetation extent. The three sites covered by the image subsets represented the range of image acquisition dates, vegetation, climate, soils and land uses across the floodplain, variables important in determining the final accuracy of image classification due to their influence on the spectral signature of landcover types.

Table 3.2: Satellite images used to map woody vegetation extent across the mid–lower Namoi floodplain.

Acquisition date	Image identification
14/11/2004	538141004111140028242B
21/01/2005	5382410050121002092J
21/01/2005	53824110501210020372J
25/11/2004	53834100411250016452J
25/11/2004	53834110411250016542J
10/11/2004	53844100411100005181J
10/11/2004	53844110411100005261J
01/01/2005	53854110501010005121J
06/05/2005	53854120505060000222J
26/05/2005	53864120505260015311J

3.2.3 Study Sites

3.2.3.1 Site 1 (Narrabri)

Site 1 was located largely on the northern side of the Namoi River 10 km east of Narrabri (30°19'S, 149°47'E). Land use in the area was mainly cropping and grazing, and included a stock route, residential subdivision, a recently harvested wheat paddock and the Namoi River. The soil type was predominantly a red-brown, silty, alluvial clay. The image (Figure 3.2) was excised from SPOT 5 multispectral data captured on 1 January 2005 (Table 3.2).

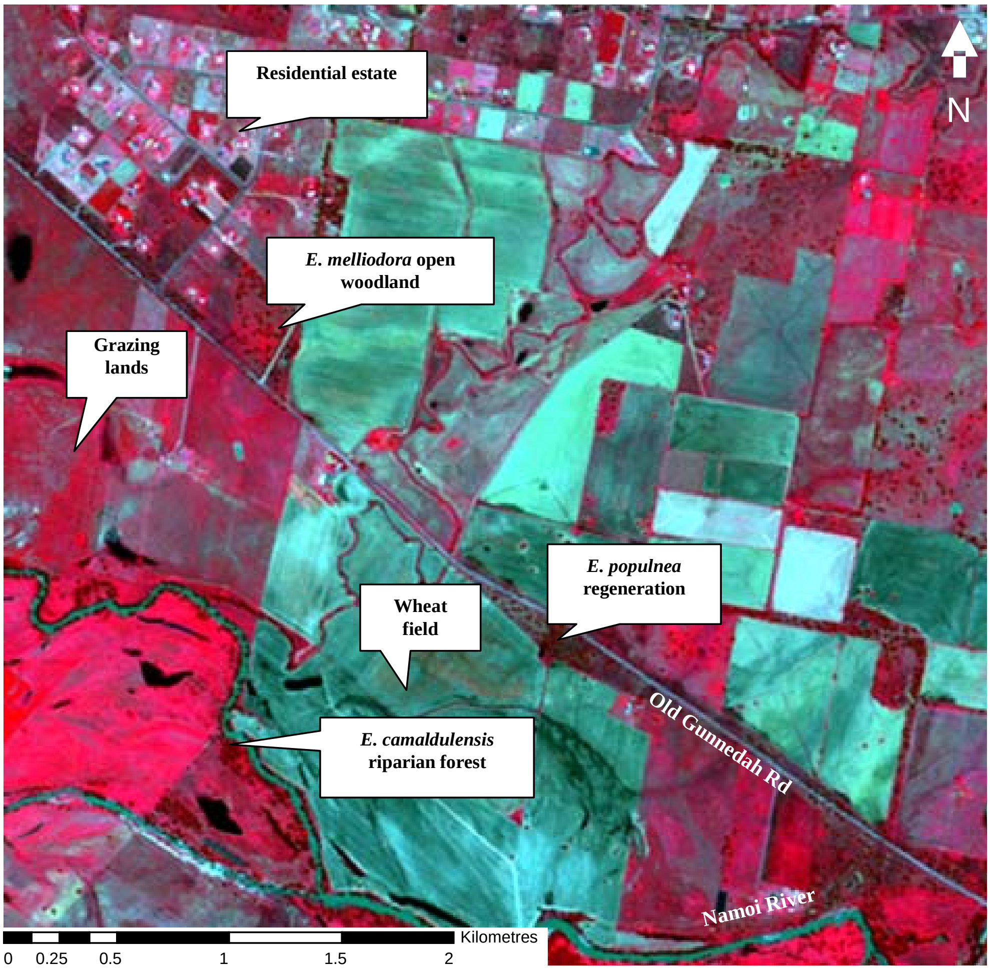


Figure 3.2: SPOT 5 image of study site 1 located 10 km east of Narrabri, NSW, indicating major landmarks, land uses and vegetation types (image date 1 January 2005).

The native woody vegetation types in site 1 included woodlands dominated by poplar box (*Eucalyptus populnea* subsp. *bimbil*), yellow box (*E. melliodora*) and river red gum (*E. camaldulensis*) (Figure 3.3). Other woody species in the site occurred as scattered individuals only. Herbaceous vegetation cover consisted predominantly of native summer-active perennial pastures.

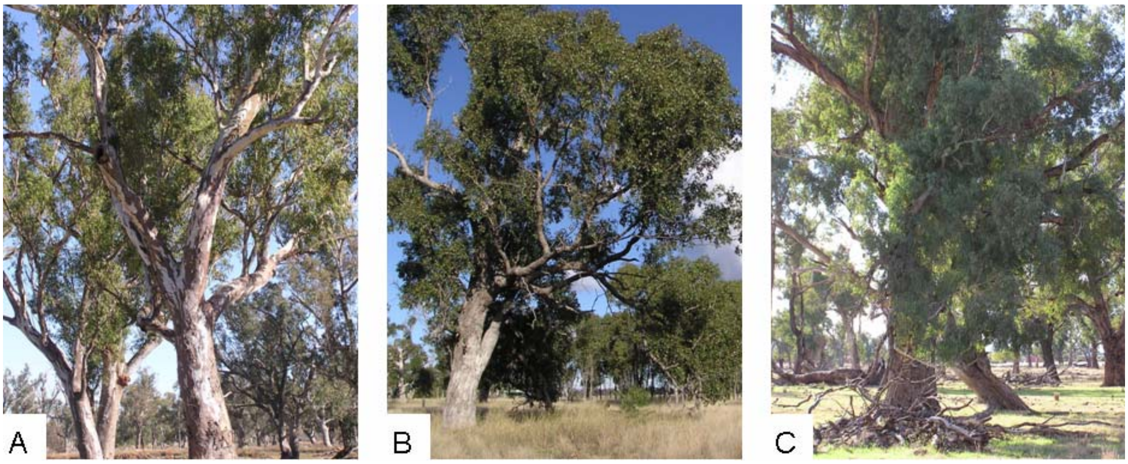


Figure 3.3: a) river red gum, b) poplar box, and c) yellow box-dominated woodlands at site 1.

3.2.3.2 Site 2 (Pilliga)

Site 2 (Figure 3.4) was 20 km north-west of Pilliga (30°21'S, 148°53'E), and was predominantly a cropping area with irrigated cotton production south of the Namoi River and dryland cereal production to the north. Native vegetation in the east of the study area was white cypress pine (*Callitris glaucophylla*) open forest. Soils varied, with red sands in the wooded, eastern part of the area, surrounded by grey clays. The date of image capture was 25 November 2004 (Table 3.2), at a time when wheat was being harvested in the north-west part of the site. *Eucalyptus, Acacia* and *Callitris* spp. dominated the native woody vegetation (DLWC 1999). Herbaceous vegetation cover was dominated by native perennial pastures, senescent turnip weed (*Rapistrum rugosum*) and black roly poly (*Sclerolaena muricata*).

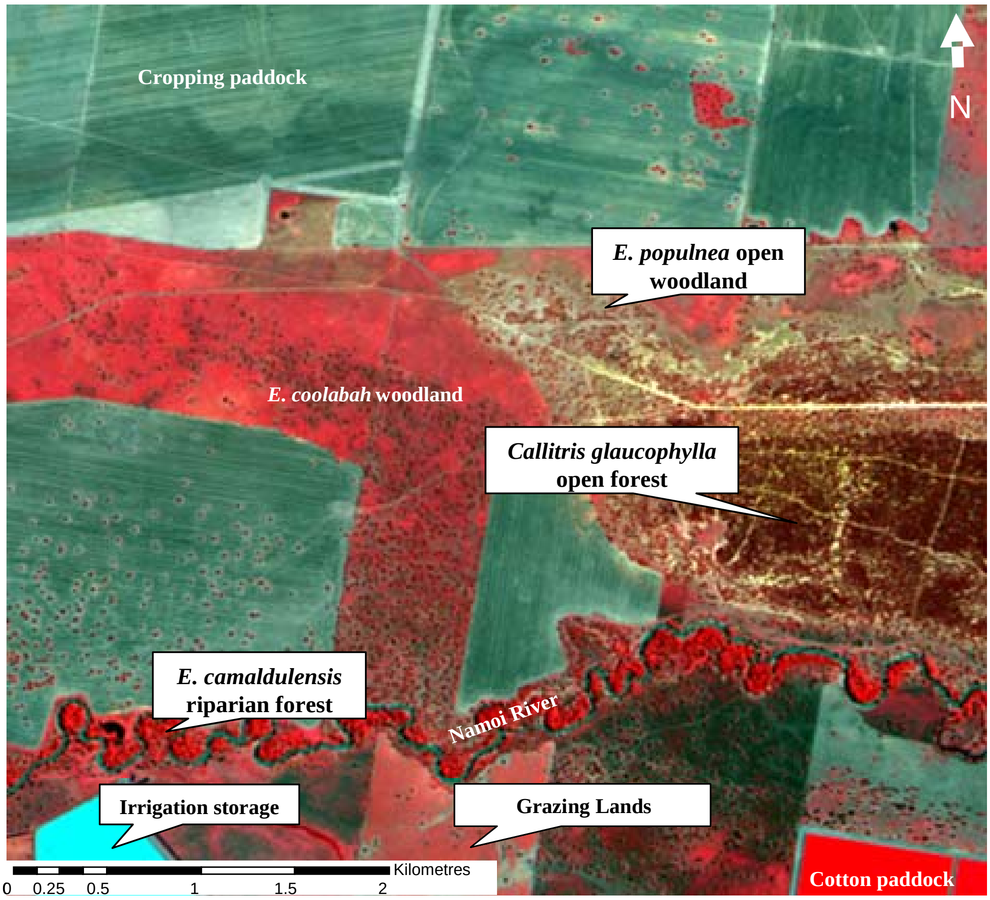


Figure 3.4: SPOT 5 image of study site 2 located 20 km north-west of Pilliga, NSW, indicating major landmarks, land uses and vegetation types (image date 25 November 2004).

3.2.3.3 Site 3 (Curlewis)

Site 3 was west of Curlewis (30°6’S, 150°16’E) on the Liverpool Plains in the upper Namoi Valley. Cropping (predominantly irrigated cotton and various cereal crops) was the dominant land use on the floodplain while the hills were wooded. Soils varied from grey and black clays on the floodplain to sandy clay loams in the hills. The date of the image (Figure 3.5) was 26 May 2005 (Table 3.2) during cotton harvest in the region. Most of the winter crops such as wheat had not been or were only just starting to be sown. Dominant native woody vegetation in the site included *Eucalyptus* and *Callitris*-dominated woodlands (Banks 1995). Herbaceous vegetation cover was predominantly native summer-active perennial pastures but winter weeds such as turnip weed were likely to have established by the time of image capture.

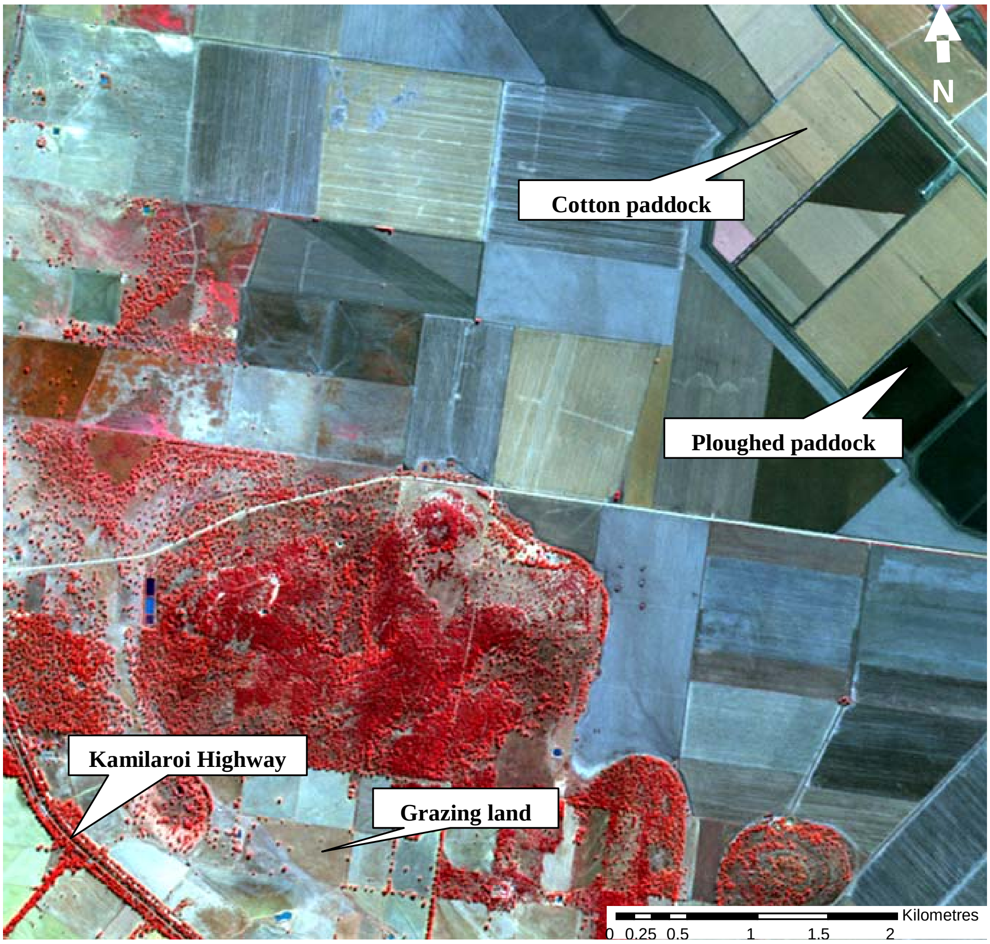


Figure 3.5: SPOT 5 image of study site 3 west of Curlewis, NSW, indicating major landmarks, land uses and vegetation types (image date 26 May 2005).

3.2.4 Image Analysis

3.2.4.1 Classification of Woody Vegetation Communities

The iterative self-organising data analysis technique algorithm (ISODATA) unsupervised classification method in ER Mapper 7.0 was used to map woody vegetation communities using 200 classes (after attempts using 30, 50 and 100 classes were unsuccessful) at site 1. ISODATA groups pixels of similar spectral characteristics into classes to be later allocated a landcover description by the user. The ISODATA method is widely used, quick and conceptually simple. Default settings were used except that the percentage unchanged (the maximum percentage of pixels whose class values are allowed to be unchanged between iterations) was set at 95%. Once the classification was complete, classes were allocated to one of the following cover class descriptions after referring to the colour composite imagery: (1) water; (2) cropping paddock; (3) grassland; (4) *E. populnea*; (5) *E. melliodora*; (6) *E. camaldulensis*, and (7) buildings.

Ten pixels for each cover category were generated from the pan-sharpened colour composite image for accuracy assessment using a stratified random sampling technique similar to the cluster sampling method explained by Jensen (2005). These points were overlayed on the unsupervised classification to determine the correspondence between each point description and the classification description. These data were used to generate an error matrix, overall accuracy, producer's accuracy, user's accuracy and Kappa statistic (Congalton 1991).

3.2.4.2 Selecting the Optimal Number of Unsupervised Classes

The ISODATA unsupervised classification technique was used to classify the multispectral data in the Narrabri, Pilliga and Curlewis sites (Figures 3.2, 3.4 and 3.5). ER Mapper 7.0 default settings were used in the classification except that the percentage unchanged was set at 95% and the maximum number of classes was set at 20, 50, 100 and 200, in turn. Classes were grouped into those representing woody vegetation, water and all other landcover classes (non-woody) using the colour composite imagery. An error matrix, overall accuracy, producer's accuracy, user's accuracy and the Kappa statistic were used to assess the accuracy of the final maps (Congalton 1991). The pan-sharpened colour composite imagery was used to check 70 points (30 woody vegetation, 30 non-woody landcover classes and 10 water) at each site, using a stratified random sampling technique.

3.2.4.3 Catchment-scale Tree Cover Mapping

The vegetation of the mid-lower Namoi Valley floodplain was mapped using ten SPOT 5 multispectral satellite images dated between 10 November 2004 and 26 May 2005 (Table 3.2). Each image was processed using the default settings of the ISODATA unsupervised classification method in ER Mapper 7.0, with the maximum number of classes set at 50 and percentage unchanged set at 95%. Classes were allocated to three categories: woody vegetation, water and all other landcover types (non-woody) through interpretation of the pan-sharpened colour composite images. Map accuracy was assessed using 1:50 000 (western end of the catchment) and 1:25 000 (eastern end) aerial photography dated between 1995 and 2003. These photographs were chosen arbitrarily, 1 from each run covering the 1:100 000 map sheets in the study region. Areas where land cover had changed (e.g. native vegetation clearance) between photograph and image capture were avoided in the accuracy assessment.

The classified images were converted to ArcGIS grid format and merged to combine the ten images into one GIS layer. Accuracy assessment was carried out by overlaying a small section of the aerial photograph with a 1 cm grid printed on transparent film. The landcover type of 80 points on the aerial photograph was compared with the same points on the classified image and agreement and errors were recorded. Some 3041 points were assessed across the study region for a reliable estimate of the accuracy of the final classification. An error matrix, overall accuracy, producer's accuracy, user's accuracy and the Kappa statistic were used to assess the accuracy of the final maps (Congalton 1991).

3.3 Results

3.3.1 Unsupervised Classification of Tree Species

Unsupervised classification was unsuccessful in mapping the three dominant tree species (*E. populnea*, *E. melliodora* and *E. camaldulensis*) separately at the study site (Table 3.3). Classification using 200 classes resulted in an overall accuracy of 79% and 75% Kappa. The accuracy assessment showed that the main tree species classes were generally confused with each other.

Table 3.3: Error matrix and accuracy assessment for unsupervised classification at site 1 using 200 classes. Columns show how many points were classified correctly and how mis-classified points were mapped.

	Grassland	*E. populnea*	*E. camaldulensis*	*E. melliodora*	Crop paddock	Water	Buildings	Row total	User's accuracy
Grassland	10	2	1	1	0	1	0	15	66.7
E. populnea	0	6	0	6	0	0	0	12	50.0
E. camaldulensis	0	1	8	0	0	0	0	9	88.9
E. melliodora	0	0	0	2	0	0	0	2	100.0
Crop paddock	0	0	0	1	10	0	0	11	90.9
Water	0	0	1	0	0	9	0	10	90.0
Buildings	0	1	0	0	0	0	10	11	90.9
Column total	10	10	10	10	10	10	10	70	
Producer's accuracy	100.0	60.0	80.0	20.0	100.0	90.0	100.0		

3.3.2 *Optimal Unsupervised Class Numbers*

Accuracy assessment of the woody versus non-woody classifications with 20, 50, 100 and 200 classes showed that there was no obvious gain in using more than 50 unsupervised classes (Figure 3.6). Overall accuracy increased from about 75% to 85% when the number of classes increased from 20 to 50. Beyond 50 classes, classification accuracy stabilised with no increase at either 100 or 200 classes. The Kappa statistic showed a parallel pattern, with approximately 50% for 20 classes rising to 65% for classifications based on 50, 100 and 200 classes.

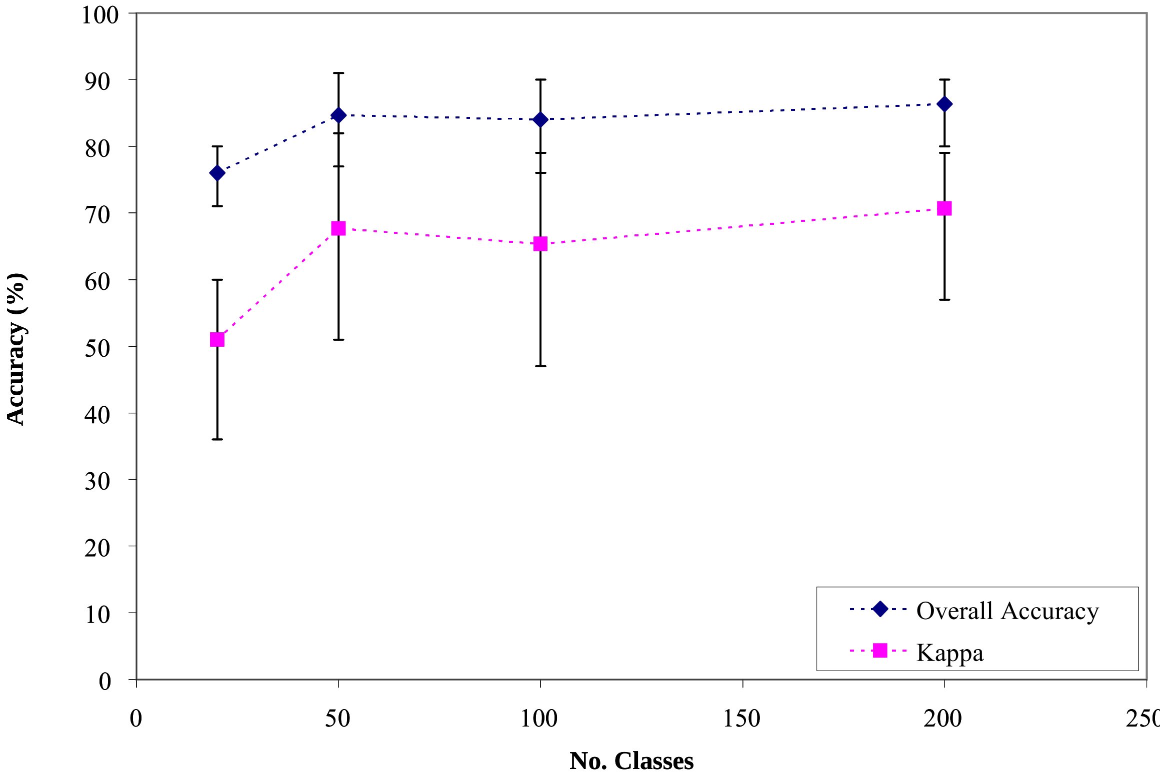


Figure 3.6: Accuracy achieved (mean and range) with increasing number of classes for woody and non-woody landcover categories. Twenty classes were insufficient to achieve a satisfactory level of accuracy but more than fifty classes did not substantially increase overall accuracy or the Kappa statistic.

3.3.3 Catchment-scale Tree Cover Mapping

Woody vegetation classification across the region achieved an overall accuracy of 89% (Table 3.4). The producer's accuracy was 53% for woody vegetation and 97% for non-woody vegetation, while the user's accuracy for woody and non-woody vegetation 77% and 90%, respectively The Kappa statistic was 56%. While water was classified, it occupied such a small area across the floodplain that none of the accuracy assessment points fell on water-covered pixels.

Figure 3.7 illustrates the extent and distribution of woody vegetation across the mid-lower Namoi floodplain as mapped using the unsupervised classification of ten SPOT 5 images. Woody vegetation covered approximately 7% of the study region while non-woody vegetation and water covered 92% and 1%, respectively. Woody vegetation was more prevalent in riparian and frequently flooded areas.

Table 3.4: Error matrix and accuracy assessment for woody vs non-woody vegetation across the Namoi floodplain. The columns show how many points were classified correctly and what mis-classified points were mapped as.

	Woody	Non-Woody	Row total	User's accuracy
Woody	285	84	369	77.24
Not woody	257	2415	2672	90.38
Column total	542	2499	3041	
Producer's accuracy	52.58	96.64		

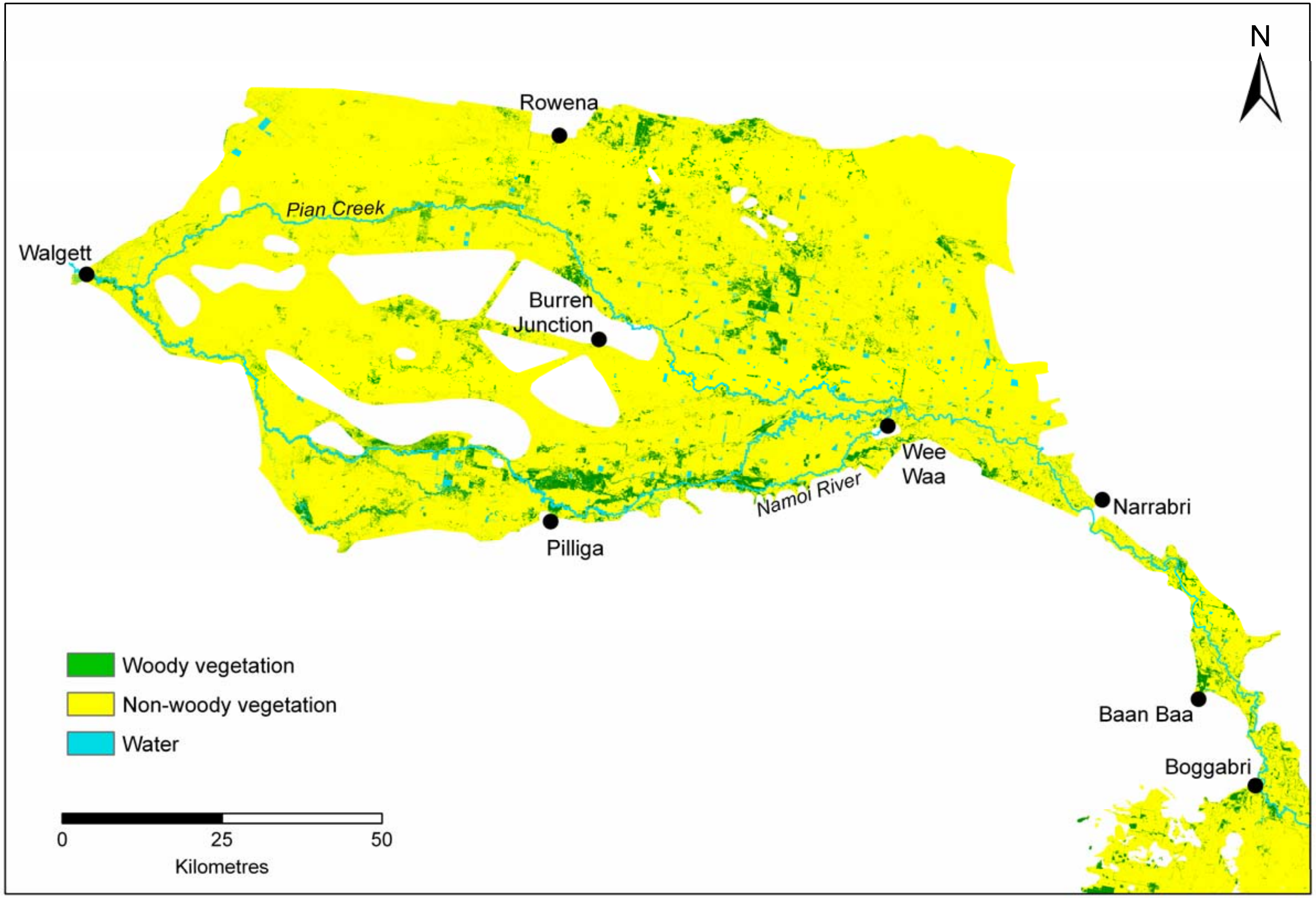


Figure 3.7: Woody vegetation of the Namoi floodplain as mapped using unsupervised classification of SPOT 5 data.

3.4 Discussion

3.4.1 Timing of Data Capture

Most studies indicate that satellite imagery acquired in summer is preferable for mapping woody vegetation cover as the differences between evergreen tree cover and cured grassland are most obvious in hot dry months (Gilbee & Goodson 1992; Kuhnell et al. 1998; Woodgate & Black 1988). However, unseasonably high rainfall or summer-dominant rainfall is likely to negate this generalisation. High rainfall in the Narrabri district in December 2004 (268.5 mm, more than five times the 100-year December average of 50 mm) demonstrated the importance of control over timing of image acquisition as rainfall patterns dictated the accuracy of the final mapping. The rain led to vigorous growth of green ground cover and resulted in confusion of herbaceous vegetation with tree cover and hence overestimation of tree cover. Swampy vegetation and water lying in paddocks were also confused with tree cover.

Mature wheat crops at site 2 had a similar reflectance to many of the senescing grasses in the grazed pastures in the image. Rainfall in the preceding month produced a flush of grass growth beneath the senescing grasses in grazed areas and a pulse of weed growth in the cultivated areas which was often confused with woody vegetation cover. Native vegetation was more easily discernible at site 3 as the image was taken in autumn when rainfall was lower in the region so summer-active grasses were senescent, wheat crops had not been planted or had not germinated and winter weeds were only just germinating.

In agriculture-dominated regions with summer-dominant rainfall, such as the Namoi Valley, the suitability of imagery capture in winter should be investigated as it is possible that dry periods in these months will allow woody and non-woody vegetation to be more easily discriminated. However, young winter crops, especially cereal crops, and winter weeds could be confused with woody vegetation and thereby hinder the versatility of imagery captured in winter.

Timing of data capture for error checking has implications for final accuracy of mapping products. In this study, the use of dated aerial photography in error checking of the catchment scale mapping exercise introduced the possibility of errors due to land clearing in the study region between the capture of aerial photographs (1995–2003) and satellite data (2004–2005). These types of errors were minimised by comparing the aerial photography with the pan-sharpened colour composite image prior to error checking and avoiding areas with obvious discrepancies.

3.4.2 Spectral, Physical and Structural Similarity of Vegetation Types

Eucalyptus, *Acacia* and chenopods have a low reflectance in the near infrared (NIR) wavelengths used to distinguish vegetation (Cox et al. 2001). Woody vegetation extent may therefore have been underestimated, particularly in *A. pendula* woodland that occurs throughout the floodplain, and *E. coolabah* woodland west of Narrabri. Both these species possess dull grey-green, sclerophyllous foliage. Ringrose et al. (1994) recorded few NIR reflective species in semi-arid and arid areas of the Northern Territory during the dry-season, and found that no regularly used vegetation indices could be applied across the climatic gradient studied. Their conclusions are pertinent to the Namoi floodplain as well with its strong decreasing east-west rainfall gradient. However, the aim of the current study was to locate woody vegetation communities for further fieldwork in the absence of more accurate data. The use of a range of band combinations as advocated by Ringrose et al. (1994) in order to accurately assess vegetation cover across a climatic gradient is not practical for projects like the present study with short time-frames and limited resources.

In the community-level mapping experiment, the accuracy of the final map was influenced by physical and structural appearance (Figure 3.3), and spectral signatures of target species. *E. populnea* and *E. melliodora* were particularly difficult to separate, due to overlap in spectral signatures of the two communities; unsupervised classification was unable to distinguish between them. *E. populnea* and *E. melliodora* communities occupy similar positions in the landscape, i.e. floodplains, river flats and minor water courses (Beadle 1981). The many herbaceous species and grasses common to both communities contributed to the similar background influence on spectral reflectance. The *E. camaldulensis*-dominated community, on the other hand, is found along major water courses such as the Namoi River (Beadle 1981). This vegetation type supports a different assemblage of understorey species and often is on a different soil type. These differences most likely made it easier to distinguish *E. camaldulensis* from the other target vegetation types.

3.4.3 Time and Increasing Class Number

While the time taken to allocate classes to landcover types was not recorded as part of this study, an increase in processing time was evident with increasing numbers of unsupervised classes. Past studies have advocated a high number of unsupervised classes (100–400) as a way of allowing for the limitations of controlled parameters (e.g. allowable dispersion around the mean and defined number of classes) on final mapping accuracy (Driese et al. 1997; Homer et al. 1997; Vogelmann et al. 1998). This approach has also been used to avoid mismatches between spectral classes and thematic classes (Lark 1995). In this study, increased class numbers did not result in greater accuracy, only increased time and therefore expenditure. Adequate accuracy for most applications was achieved with a reduced number, in this case 50 unsupervised classes.

3.4.4 Implications for Environmental Management

Remnant vegetation in fragmented rural landscapes is important in maintaining ecological functioning and ecosystem service benefits for agriculture and the wider community, and must be managed to ensure its persistence in the landscape. With the pressures of an increasing human population and the uncertainty of climate change, better management of vegetation resources is vital to minimise biodiversity loss and maximise ecosystem service provision and human well-being. Appropriate management of vegetation assets is predicated on accurate identification of vegetation or community types and their extent and condition. There is a paucity of this basic information in many parts of the world. Filling this knowledge gap is crucial for appropriate vegetation management, restoration and rehabilitation.

3.5 Conclusions and Future Research

The mapping reported here allowed woody vegetation to be identified with a high degree of confidence in the absence of more reliable data. A quick and moderately accurate method for mapping woody vegetation has been described and limitations of the method highlighted. The research suggests that single-date SPOT 5 satellite data has value in mapping the extent of woody vegetation quickly with a high degree of accuracy, but is of little use in mapping vegetation communities. The imagery for this project had little value in mapping individual tree species, but directed image capture and multiple-date data in future projects may help overcome spectral overlap issues between vegetation types due to rainfall variability and plant phenology.

Chapter 4: Carbon Storage Value of Native Vegetation on the Lower Namoi Floodplain

A stately river red gum tree (*Eucalyptus camaldulensis*) with diameter over 3 m on the edge of the Namoi River near Narrabri, January 2008.

Chapter 4: Carbon Storage Value of Native Vegetation on the Lower Namoi Floodplain

4.1 Introduction

Accelerated global climate change is one of the most pressing environmental issues facing the world today (IPCC 2007). Predicted scenarios resulting from climate change entail varying degrees of impact on human well-being through changes in sea level, fresh water availability, human health and agricultural production, and subsequent global water and food shortages (MA 2005; Stokes & Howden 2008, 2010). Mass extinctions and severe biodiversity loss are probable as a result of climate change, resulting in altered ecosystem composition, structure and function and flow-on impacts on ecosystem service provision (Fischlin et al. 2007; MA 2005).

Increased greenhouse gas (GHG) concentrations in the atmosphere, especially of anthropogenic origin and as a result of human-induced land use change (i.e. conversion of forested lands for agricultural purposes), are the primary cause of accelerated climate change (Crowley 2000; Houghton 1999). Carbon dioxide (CO_2) is one of the most important GHGs and in 2004 accounted for 77% of total anthropogenic GHG emissions (Olivier 2005). While other GHGs such as CH_4, O_3, N_2O and NO_x are more potent on a per molecule basis in terms of contribution to the enhanced greenhouse effect, CO_2 far exceeds other GHGs in importance due to the large increase in its concentration in the atmosphere in recent decades (Rodhe 1990).

The Kyoto Protocol came into force on 16 February 2005 and commits most ratifying countries to reduce GHG emissions using 1990 emissions as a baseline (Breidenich et al. 1998). Two options exist for reducing atmospheric CO_2 concentrations and hence slowing global climate change: (1) decreasing emissions, and (2) increasing C sequestration. While reductions in fossil fuel emissions are more likely to result in a decrease in atmospheric CO_2 concentrations, vegetation management, through its influence on both vegetation and soil, provides one of the best opportunities for increased C sequestration in the short term (Batjes 1998; House et al. 2002; Prentice et al. 2001). Ideally, CO_2 emissions should be in equilibrium with the capacity of natural systems to sequester C, thereby resulting in no net increase in atmospheric CO_2 concentrations. Currently, CO_2 emissions exceed the ability of natural systems to sequester C and so atmospheric concentrations of CO_2 are rising.

4.1.1 Carbon Storage as an Ecosystem Service

Carbon sequestration is an important regulating service in terms of maintenance of a breathable atmosphere (MA 2005). Vegetation, in conjunction with soils, store large quantities of C. Approximately 34% of the estimated annual CO_2 emitted through land use change and fossil fuel emissions is absorbed by terrestrial C sinks (House et al. 2002). Oceanic sinks absorb an additional 26% of anthropogenic GHG emissions, while the rest (40%) enriches the atmosphere. These figures are not static: both emission and sequestration rates are spatially and temporally dynamic due to both natural and human-induced causes (Brown 2002; Sitch et al. 2003).

Carbon storage by a terrestrial ecosystem, in the first instance, is dictated by climate (season length, rainfall and temperature), soil fertility, atmospheric CO_2 concentration, and plant functional type (leaf area, photosynthetic pathway—C_3, C_4 or CAM) (Chapin et al. 2002). These factors determine rates of photosynthesis and biomass production (gross primary production, GPP). Decomposition and respiration rates influence the ability of an ecosystem to accumulate biomass (stored chemical energy) and C. Rates of decomposition are influenced by the physio-chemical environment (climatic variables including temperature and moisture availability, and soil variables including pH, chemistry, texture and parent material), resource quality (chemistry and palatability to decomposer organisms) and the decomposer community (size of the organisms, resource specificity) (Swift et al. 1979). Respiration rates are determined by photosynthesis and decomposition rates, with the additional consideration of herbivore interactions. Figure 4.1 summarises the major C fluxes in a terrestrial ecosystem.

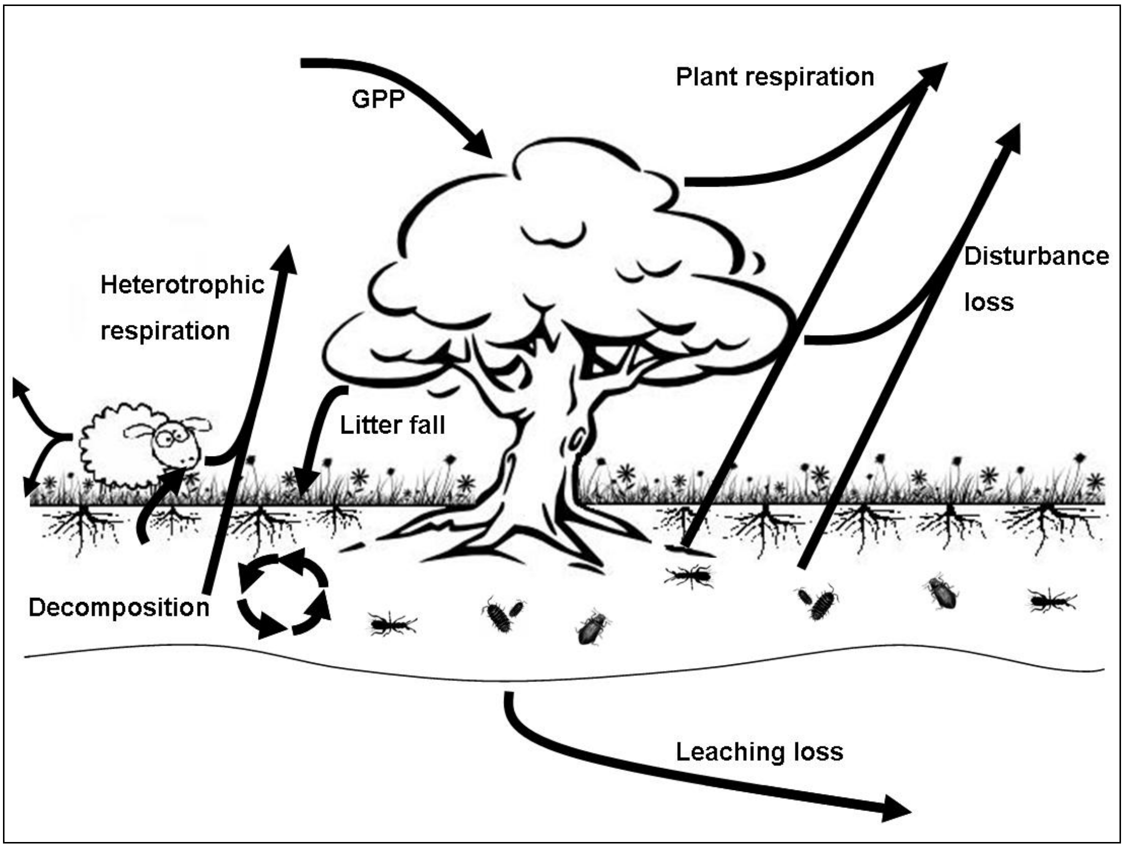


Figure 4.1: Major C fluxes in an ecosystem. The net accumulation of C in an ecosystem is equal to GPP less losses due to plant and heterotrophic respiration, disturbance and leaching (adapted from Chapin et al. 2002).

Approximately half of the C entering an ecosystem during photosynthesis is respired by plants and returned to the atmosphere; the remaining C is stored in plant tissues. The largest releases of C in an ecosystem occur between plants and the soil through litterfall and root exudation, giving rise to soil organic matter (SOM), though a percentage is eaten and respired by animals, and losses may occur due to ecosystem disturbances (Chapin et al. 2002). Litter is decomposed by micro-organisms in the soil. Again a large proportion is respired, and the rest is converted to less decomposable fractions (such as humus). Some of the soil C is leached from the ecosystem, or transferred through erosion or movement of animals.

4.1.2 Vegetation Condition for C Storage

Photosynthesis, decomposition and respiration can be enhanced or depressed through natural or anthropogenic disturbance, thereby allowing stochastic or deliberate manipulation of ecosystem C storage (Schimel et al. 2001). Vegetation condition in a C sequestration context is determined by the ability of an ecosystem to maximise photosynthesis, store C fractions and minimise disturbances that increase decomposition and respiration rates. Several components of terrestrial ecosystems need to be accounted for in estimates of C storage value including living and dead biomass and soil C. In forest ecosystems, the dominant C stores are in woody vegetation including roots, and soil (Brown 2002). In some forest ecosystems litter, dead standing wood and coarse woody debris (CWD) can also be significant (Brown 2002), representing up to 20% of the total C stored in the system (Delany et al. 1998). In savanna ecosystems, soil C is the most C dense component representing approximately 80% of the ecosystem C store (Scholes & Hall 1996).

4.1.2.1 Above-ground Woody Vegetation

Woody vegetation, where it occurs in terrestrial ecosystems, comprises the majority of the biomass (weight of plant material) and is therefore an important constituent in terms of C storage. Contribution of woody vegetation to soil C storage may be disproportionately larger than herbaceous vegetation not only in terms of the amount of C contributed, but also in residence time: woody vegetation inputs to soil contain more lignin and have higher C:N ratios, making it less accessible to decomposer organisms and increasing residence time in the soil (Lavelle & Spain 2001; Spain et al. 1983). Structurally diverse vegetation including woody plants stores substantially more C in both soils and vegetation than agriculturally derived grasslands (Eldridge & Wilson 2002; Wilson et al. 2009; Young et al. 2009).

While the value of woody vegetation for C sequestration is largely accepted, the value of forests of different ages is less clear. Until recently, there was a perception that forests decrease in C sequestration value with age and that old growth forests were C neutral and thereby redundant in the global C cycle (Zhou et al. 2006) and that efforts to reduce CO_2 concentration in the atmosphere should be directed towards younger, faster growing forests and regrowth. Indeed, this is a basic assumption of the Kyoto Protocol (Luyssaert et al. 2008; Schulze et al. 2000). This assumption arose from studies such as those by Odum (1969) that noted a decrease in net primary production (NPP) in mature communities. However, these studies concentrated on single-age, single-species experimental communities. These conclusions do not necessarily hold true for natural systems due to the presence of multi-aged individuals and a diversity of species characterised by differing growth rates (Carey et al. 2001). Luyssaert et al. (2008)

reported that net ecosystem productivity (NEP) in forests aged 15 to 800 years is usually positive, indicating a CO_2 sink, and in a range of natural old growth forests of differing species composition and growing conditions, biomass accumulation continued with increasing stand age for centuries. Harmon et al. (1990) noted that it is not the annual uptake of C that is important, but the amount of C stored in the forest.

Conversion of old growth forests to younger, faster growing forests results in large emissions of CO_2 to the atmosphere. In addition to loss of actual woody biomass (a reduction of approximately 350 Mg per ha from an old growth forest), soil organic C (SOC) decreases, due to intensive management, and the site may continue to lose C for more than three decades after disturbance (Harmon et al. 1990). Based on reasoning, old growth forests form a high quality benchmark for condition in terms of C sequestration, and are more valuable than young forests and plantations (Harmon et al. 1990; Luyssaert et al. 2008).

4.1.2.2 Above-ground Herbaceous Vegetation

Herbaceous vegetation in wooded ecosystems constitutes only a very small portion of the total C pool. However, in grassland and savanna ecosystems, herbaceous vegetation is the primary source of biomass, though the total biomass in the system at any particular time is much smaller than in woodland ecosystems (Lloyd et al. 2008). The majority of NPP in grassland ecosystems occurs belowground, though the exact amount is difficult to measure and often underestimated (Gibson 2009). Root exudates in particular are frequently unaccounted for and despite careful sampling procedures, the fine root component is often lost or underestimated (Redman 1992). Coupland (1993) reported that the proportion of NPP occurring underground in grasslands across the world varied between 22% in a mixed grass community in Pilani, India, and 81% in an *Andropogon* community in Sambalpur, India, though exactly half the sites included had belowground NPP ≥ 50%.

In wooded ecosystems, herbaceous vegetation productivity is influenced by competition (negative relationship) or complementarity (positive relationship) with the woody component and the litter it produces. Biotic factors influencing herbaceous vegetation productivity in these ecosystems include (Scholes & Archer 1997): (1) competition for water and nutrients between the root systems of the trees and grasses; (2) reduction of the photosynthetically-active radiation and rainfall reaching the grass canopy, through interception by the tree canopy; and (3) improved growing conditions for the grasses immediately below the tree canopy due to local nutrient enrichment and improved soil water conservation. Abiotic influences on herbaceous vegetation productivity can be positive or negative, physical or chemical in nature and include (Xiong & Nilsson 1997): (1) barriers to germination and emergence of seedlings through reduced access to resources such as mineral soil, light and radiation; (2) protection of seedlings from predation, erosion, adverse temperature or moisture extremes; (3) impacts on soil fertility and mineral composition; (4) phytotoxins; (5) control of decomposer food webs and habitat for fauna and microbes, and (6) creation of bare ground for germination through suppression of existing vegetation.

In general, larger tree basal area results in lower grass productivity (Jackson & Ash 1998: Scanlan & Burrows 1990; Scholes 2003; Scholes & Archer 1997). However, some studies have found that trees can have a positive influence on herbaceous biomass production (e.g. Scholes & Archer 1997; Weltzin & Coughenour 1990). The impact of trees depends on the ecophysiological or specific characteristics of the tree and grass growth forms (canopy architecture, rooting patterns), photosynthetic pathway, photosynthetic habit (evergreen or deciduous) and resource requirements (light, water and nutrients), as well as availability of resources (determined by seasonality, precipitation etc.), grazing and other disturbances (Scholes & Archer 1997). Tree removal studies have shown that increased grass productivity cannot be sustained indefinitely after tree removal (Jackson & Ash 1998). It is likely that maximum herbaceous vegetation biomass production is achieved in mixed tree and grass systems (Scholes & Archer 1997; Scholes 2003)

4.1.2.3 Litter

Litter accumulation can be substantial in some vegetation communities, but this component of the biomass pool is often neglected in C accounting studies (McKenzie et al. 2000). Litter is defined as dead, unattached plant material including leaf litter, twigs, flowers, fruits, seeds and dead micro-organisms. Litter accumulation in ecosystems is determined by the difference in production (litterfall), and decay (decomposition) rate or loss from the system (Birk & Simpson 1980).

Compared to surrounding ecosystems, litter production and decomposition are greater in riparian zones where NPP is generally greater and conditions are favourable for an active soil food web (Xiong & Nilsson 1997). Mature forests, in general, have much higher litterfall rates than early successional vegetation (Naiman et al. 2005). In eucalypt forests, the composition of the litter varies from leaf-dominated (90% leaf) in young stands to wood and bark-dominated in older stands (20% leaf in old stands) (Snowdon et al. 2005). A corresponding increase in C:N ratio occurs with the shift from leaf to woody material in litterfall, which influences decomposer food webs, and in turn has implications for decomposition and respiration rates (Lavelle & Spain 2001). Litter with a high C:N ratio is generally slower to decompose, but rates depend on site conditions. Decomposition of low-quality litter (high C:N ratio) can be increased in the presence of high-quality litter (low C:N ratio) (Spain & Lavelle 2001), such as that contributed by nitrogen-fixing *Acacia* species.

4.1.2.4 Coarse Woody Debris (CWD) and Dead Standing Wood

CWD is distinguished from litter by size, although no consensus has been reached on the boundary between litter and CWD (Harmon & Sexton 1996). The general range in CWD extends between 1 cm (e.g. McGee et al. 1999) and 25 cm (e.g. Bingham & Sawyer 1988); the Australian standard is >2.5 cm (McKenzie et al. 2000). Substantial amounts of C can be stored in CWD, especially in species with a high resistance to decay, where decomposition may take hundreds of years (Mackensen et al. 2003). CWD of river red gum (*Eucalyptus camaldulensis*), for example, has a lifetime of 375 years (Mackensen et al. 2003).

In addition to ambient conditions (e.g. temperature and climate), decomposition of CWD differs with each piece of wood according to temperature of the decaying log, moisture content, wood density, size, species, C:N ratio, and gas and water exchange rates (Harmon et al. 1986; Mackensen et al. 2003). Temperature and moisture content are influenced by external factors (climate, site microclimate etc.), and internal factors such as log diameter, surface:volume ratio (access for decomposer organisms) and bark cover, log position and shading. Some species also contain substances toxic to decomposer organisms. For example, coolibah (*Eucalyptus coolabah*) is reported to be termite resistant (J. Moore, pers. comm., May 2007).

Dead standing wood is often included in the CWD component in C accounting studies, but lack of contact with the soil induces slower decomposition rates compared to fallen CWD. Reduced access for decomposer organisms and suboptimal conditions for decomposition contribute to slower decomposition rates (Yatskov et al. 2003). The biomass of dead standing wood is determined by natural (e.g. drought, disease, herbivory, natural senescence, natural disturbance) and management-related factors (e.g. ringbarking, poisoning, fire, induced stress), and decomposition rates.

4.1.2.5 Soil and Plant Roots

The global soil C pool is estimated to be about 2500 Gt, approximately 3.3 times the size of the atmospheric C pool, though great variability in soil C stores exist (Lal 2004). Soil C storage is therefore an important consideration in C accounting (Jobbágy & Jackson 2000).

The vertical distribution of C in an ecosystem varies with vegetation and soil type. Analysis of root to shoot ratios in different biomes shows that vegetation types apportion biomass differently according to species characteristics, soil moisture and nutrient availability, regeneration strategies and competition for light (Mokany et al.

2006). Carbon distribution in the soil profile also differs with vegetation type. Litterfall is one of the major pathways for energy and nutrient transfer between forest and soil (Chapin et al. 2002). Forested ecosystems store C closer to the soil surface compared to grasslands and shrublands that have higher C concentrations at depth (Jobbágy & Jackson 2000). Such generalisations about C distribution in soil may not apply to vertosols, as organic matter can be incorporated into deeper layers of the soil during wetting and drying cycles and the associated formation of deep cracks and soil churning (Baldock & Skjemstad 1999; Hubble 1984; Jobbágy & Jackson 2000; Young et al. 2005).

Soil organic C includes plant, animal and microbial residues in varying stages of decomposition as determined by complex interactions between chemical, physical and biological processes (Post & Kwon 2000). Isotope analysis has shown that C in soil has a residence time of 200–2220 years, when examined across a range of ecosystems, soil types and climatic conditions (Swift et al. 1979). The greatest inputs of C to the soil are of plant origin, from litterfall (discussed previously), or from root systems through exudation, secretion, sloughing and lysis of cells, and senescence of root tissue (Rees et al. 2005).

Soils are significant sinks for CO_2, but depending on land management practices, soils also have the potential to be CO_2 sources, especially where inputs from photosynthesis are less than respiratory losses. These conditions are prevalent when ecosystems are disturbed, such as when new agricultural systems are created from newly cleared arable land (Schimel et al. 2001). The initial loss of soil C after disturbance can be substantial, but differs with soil type (Dalal & Chan 2001). Hassink et al. (1997) showed that fine-textured soils provide greater protection of C in the clay and silt fractions of the soil when cultivated, thereby allowing fine-textured soils to accumulate greater quantities of C and for losses to be reduced compared to other soils (Dalal & Chan 2001). This may be due to the ability of organic matter in soils to associate with clay and silt particles to form clay–organic complexes (Coulombe et al. 1996).

Plant life cycles significantly influence SOC concentrations through their impact on root growth and senescence (Rees et al. 2005). For example, annual species generally have high initial root growth, after which root production stops or is reduced. In contrast, perennial plants tend to replace rather than maintain individual roots, perhaps because it is less expensive in terms of energy to produce a new root than to maintain an existing one. Perennial vegetation thereby provides greater root-derived C inputs to the soil.

4.1.3 Research Gaps

Measurement of C storage and sequestration is currently being undertaken at all scales, from individual soil aggregates to whole continents, and in a wide range of ecosystems, from deserts to tropical rainforests. Countries that have ratified the Kyoto Protocol, in particular, are seeking information on both emissions and sequestration potential for C accounting purposes. Research into C sequestration and storage has generally centred on higher rainfall zones, especially tropical areas, with little consideration given to the importance of semi-arid rangelands (Pacala et al. 2001). Data on C sequestration in semi-arid woodlands is lacking in Australia (Keith et al. 2000), which is surprising considering these ecosystems cover a large portion of the Australian land surface and small changes to land management practices have the potential to sequester large amounts of C.

4.1.4 Aims and Objectives

The aim of this study was to quantify the C store of six common vegetation types of the lower Namoi floodplain. The objective was to determine differences in C storage for multiple ecosystem components and vegetation types, and hence determine their value in providing a C storage service.

4.2 Methods

4.2.1 Study Region

The lower Namoi floodplain in northern NSW, Australia (Figure 4.2), is dominated by agriculture. The study region has a semi-arid subtropical climate with unreliable and sporadic rainfall throughout the valley. Mean annual rainfall varies from 400 mm near Walgett in the west to 600 mm near Boggabri in the east, with a slight summer dominance (Stannard & Kelly 1977). Mean maximum summer temperatures increase slightly across the study area from east to west. In summer, mean maximum temperatures range between 33–36˚C, while winter mean minima are 3–4˚C (BOM 2008).

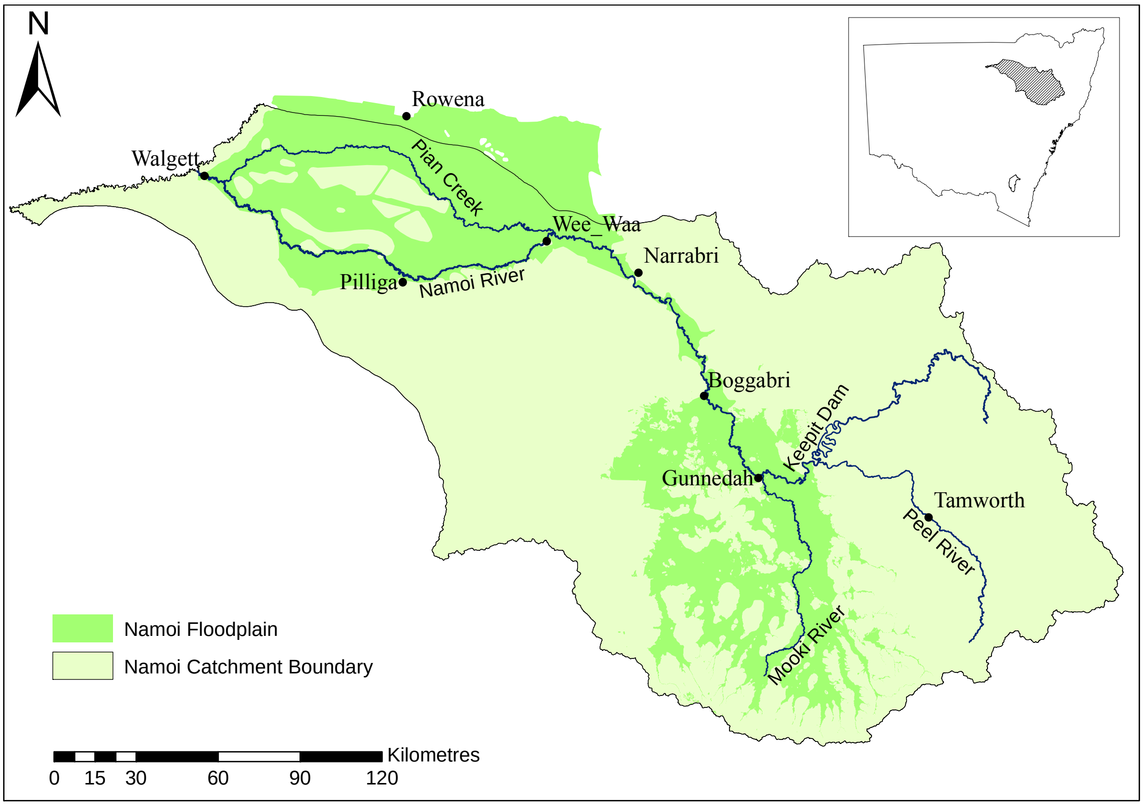


Figure 4.2: Namoi floodplain in northern NSW, Australia. The study region was the lower Namoi floodplain from Boggabri downstream to Walgett.

Soils across the study region are mainly Vertosols (Isbell 1996), mostly black, grey or brown clays, typically with self-mulching characteristics, slightly to strongly alkaline, and with free $CaCO_3$ at varying depths (Stannard & Kelly 1977). Vegetation patterns on the floodplain are largely dictated by the presence and movement of water across the landscape (Kearle et al. 2002). In general, on the cracking clay soils, woody vegetation occurs in lower lying areas and along drainage lines, while large expanses of natural grassland occur in drier, less frequently flooded areas. Land use history and management of native vegetation varies across the region. Areas of coolibah and black box were ring-barked in the 1900s (J. Moore, pers. comm., 2006) dead standing trees still remaining where that vegetation type occurs around Walgett, Pilliga and Burren Junction. Some stands of coolibah and black box were allowed to coppice after being ring-barked and now exist as stunted, multiple-stemmed trees. In other areas, timber has been clear-felled or thinned to encourage pasture growth, allow cropping, or cut for firewood and fence posts. Approximately 7% of the region is woody vegetation (Chapter 3), while large areas of derived or natural grasslands, usually grazed, are interspersed among cropping paddocks.

Sixty-one sites were located on cotton farms and travelling stock routes on the lower Namoi floodplain, between Boggabri (–30.7000˚, 150.0333˚) in the east and Walgett (–30.0167˚, 148.1167˚) in the west. Six vegetation types commonly occurring in the region were targeted: (1) river red gum-dominated gallery forests and woodlands; (2) coolibah woodland; (3) myall (*Acacia pendula*) woodland and open woodland; (4) black box (*Eucalyptus largiflorens*) woodland; (5) native tree and shrub plantings; and (6) native and derived grasslands. Sites were chosen to represent the range of recent management histories. A balanced design could not be achieved owing to lack of availability of sites in some vegetation types subject to certain kinds of management.

4.2.2 Field Sampling

4.2.2.1 Above-ground Woody Vegetation Biomass

Biomass was sampled in quadrats stratified by vegetation type. Quadrats varied in size (25 × 25 m to 1 ha) according to tree density. Diameter of trees was measured at breast height (1.3 m) over bark (DBHOB) using a metal diameter tape or electronic digital calipers. For trees or shrubs where this measurement was not possible or sensible due to low branching habit or diminutive height of the specimen, height was measured using height poles, or the diameter at 30 cm above ground level ($diam_{30}$) was measured. Tree

DBHOB was measured with 0.1 cm accuracy and height was measured to the nearest 0.1 m. Dead standing material was also assessed in this manner. Where the dead tree was reduced to no more than a hollow stem, biomass was calculated using a method similar to that used for estimation of CWD biomass.

Allometric equations were used to estimate tree biomass based on tree DBH, diam_{30} or height. The equations used were developed by Snowdon et al. (2000) and are used by the Australian Greenhouse Office (AGO). Five equations were used for shrubs, woodland trees (DBH and diam_{30}), native plantations and native sclerophyll forest, respectively (Snowdon et al. 2000):

$$\text{Ln(biomass)} = -1.0668 + 2.8807\ (\ln(\text{shrub height})) \qquad \text{Equation 1}$$

$$\text{Ln(biomass)} = -1.4481 + 2.2364\ (\ln(\text{DBH})) \qquad \text{Equation 2}$$

$$\text{Ln(biomass)} = -2.2268 + 2.4190\ (\ln(\text{diam}_{30})) \qquad \text{Equation 3}$$

$$\text{Ln(biomass)} = -2.2450 + 2.3582\ (\ln(\text{DBH})) \qquad \text{Equation 4}$$

$$\text{Ln(biomass)} = -1.9335 + 2.3501\ (\ln(\text{DBH})) \qquad \text{Equation 5}$$

4.2.2.2 Litter and Above-ground Herbaceous Vegetation Biomass

Litter and herbaceous biomass was sampled in autumn and spring 2008 using a modified BOTANAL (Tothill et al. 1978) method. Twenty calibration quadrats representing the range of conditions and species to be encountered at three locations (near Walgett, Wee Waa and Narrabri or Boggabri) across the floodplain were harvested in each sampling period. Litter was harvested by hand and stored separately prior to harvest of standing biomass 2 cm above ground level using secateurs. Litter was defined as dead, unattached plant material. Harvested biomass was sorted into green and dead material and these samples along with the litter samples were dried for 72 hours at 60˚C. Dead material, litter and green material were weighed separately and the percent green leaf by weight, total herbage mass and litter mass calculated. For each observer, regression relationships were developed to establish relationships between actual and estimated values for percent green leaf, litter and herbaceous vegetation biomass for the calibration quadrats. These regression relationships were used to predict percent green leaf, total herbage and litter for quadrats at each site. Calibration quadrats were scored on a daily basis to correct for day-to-day variation in observers' estimates of biomass.

Litter, total herbaceous vegetation biomass and percent green leaf were scored in twenty 50 × 50 cm quadrats located at 4 m intervals around the perimeter of a 20 × 20 m quadrat used for floristic survey at each site. Weather conditions in the months preceding sampling were favourable for maximum herbaceous vegetation growth, therefore the data estimate the maximum vegetation biomass potential at sites under prevailing management. In converting biomass dry weight to mass of C, it was assumed that 50% of the herbaceous vegetation biomass was C (Snowdon et al. 2000).

4.2.2.3 CWD

The length and end diameters of each piece of CWD were taken using calipers and a metal tape in order to generate a volume. The percent missing of each piece was estimated, and the source species (where possible) and its status as sound or rotten was recorded. Samples of sound and rotten wood were collected across the study area for river red gum, coolibah and myall and the average density used to convert volume to mass. Where CWD crossed the quadrat boundary, an imaginary cut was put across the specimen so only wood that was inside the quadrat was included. Quadrat size varied between 10 × 10 m for the site with the largest amount of CWD to 1 ha at more open sites.

4.2.2.4 Soils

Soil C was sampled using a manual soil-coring device as per current standards (McKenzie et al. 2000, 2002), with soil cores divided into four depth increments, 0–5 cm, 5–10 cm, 10–20 cm and 20–30 cm. A depth of 30 cm was chosen to be consistent with soil C sampling protocol (McKenzie et al. 2000; McKenzie et al. 2002). At each site, a quadrat 25 × 25 m was located in a representative area of the vegetation type and management history of interest. Where possible, sites were located at least 30 m from boundaries with other land uses and different management histories. Nine cores were collected at each site, stratified according to the cover of trees, shrubs, herbs, grasses, litter and bare ground. Cores were stored in zip-lock plastic bags in cool conditions for up to 1 week and then stored at 4°C for up to 2 weeks prior to air drying.

Soil samples were bulked by depth at each site and ground using a mechanical grinder to pass a 2 mm sieve. Moisture content was determined on a subsample (after drying at 105˚C for 48 hours) for subsequent calculation of oven dry bulk density. Another subsample was crushed and passed through a 0.5 mm sieve for analysis of percent total organic C (TOC) content using a Carlo Erba NA 1500 Solid Sample Analyser. Where soil pH > 7.5, a subsample treated to remove carbonates using 2% orthophosphoric acid was measured immediately after the corresponding untreated subsample. Where treatment to remove carbonates had no effect on the percent C measured, the average percent C value of the treated and untreated samples was used in further analysis. Where treatment removed carbonates, the percent C value of the treated sample alone was used.

4.2.2.5 Roots and Total Site C

Root biomass was estimated using recommended root to shoot ratios: 0.25 × above-ground biomass for woody vegetation (Snowdon et al. 2000) and 0.5 × above-ground vegetation for herbaceous vegetation (Mooney 1972). Total site C was determined by summing the C content of each component and expressed on a per ha basis.

4.2.3 Statistical Analysis

Parametric analysis of variance (AOV) in Statistix 8 (Analytical Software 2003) was used to examine differences in C contribution by woody vegetation, herbaceous vegetation, litter, CWD, dead standing wood, roots, soil and total site C for each of the six vegetation types. Data were transformed to achieve normality; where normality of data could not be achieved using log or square root transformations, rank-transformed data were used. Significant differences between vegetation types for each C component were determined using least significant difference (LSD). Spearman rank correlations were generated to highlight relationships between vegetation characteristics and C contribution in woody vegetation, herbaceous vegetation, litter, CWD, dead standing wood, and soil across all vegetation types. Spearman's correlations were used as data was not normally distributed. Differences were regarded as significant at $P \leq 0.05$.

4.3 Results

4.3.1 Above-ground Woody Vegetation

The mean ($F = 6.52$, $P < 0.001$) and maximum ($F = 12.2$, $P < 0.001$) DBH of the dominant species at each site varied between vegetation types (Table 4.1). River red gum sites had the greatest mean and maximum DBH values: both these values were at least twice size of the next largest species, coolibah. Myall and black box sites had similar mean and maximum DBH values, which were considerably (but not significantly) less than coolibah. Where grassland sites had trees present, these were relatively large. The number of trees at each site was greatest in planted sites, and lowest in grasslands ($F = 12.6$, $P < 0.001$).

Table 4.1: Site mean and maximum DBH of the dominant species by vegetation type (note: all species used in grassland and planted sites).

	Number of sites	Mean DBH (cm)	Max DBH (cm)	Mean number of trees at sites
River red gum	13	68.50±10.89a	217a	99.54±37.32b
Coolibah	17	28.76±5.09b	104b	318.12±149.32ab
Myall	6	17.83±5.23b	61b	295.67±163.77ab
Black box	4	19.75±3.84b	67b	116.00±49.59ab
Grassland	17	36.13±6.57ab	74b	1.18±0.96c
Planted	4	11.75±1.89b	29b	508.00±46.01a

River red gum and coolibah-dominated vegetation had the highest amounts of above-ground woody biomass of the six vegetation types, while grasslands had very little woody biomass (Figure 4.3). Myall, black box and planted vegetation had intermediate quantities of above-ground woody biomass. The river red gum – coolibah group had a mean of 72.2 t C ha^{-1}, the myall – black box – planted group 15.0 t C ha^{-1} and the grassland group 1.1 t C ha^{-1}. While river red gum and coolibah sites were included in the same group (after rank transformation), river red gum sites had more than twice the C stored in above-ground woody vegetation, on average, compared to coolibah sites. The largest quantity of C in above-ground woody vegetation was 235.9 t C ha^{-1} at a river red gum site on a stock route adjacent to the Namoi River, east of Narrabri.

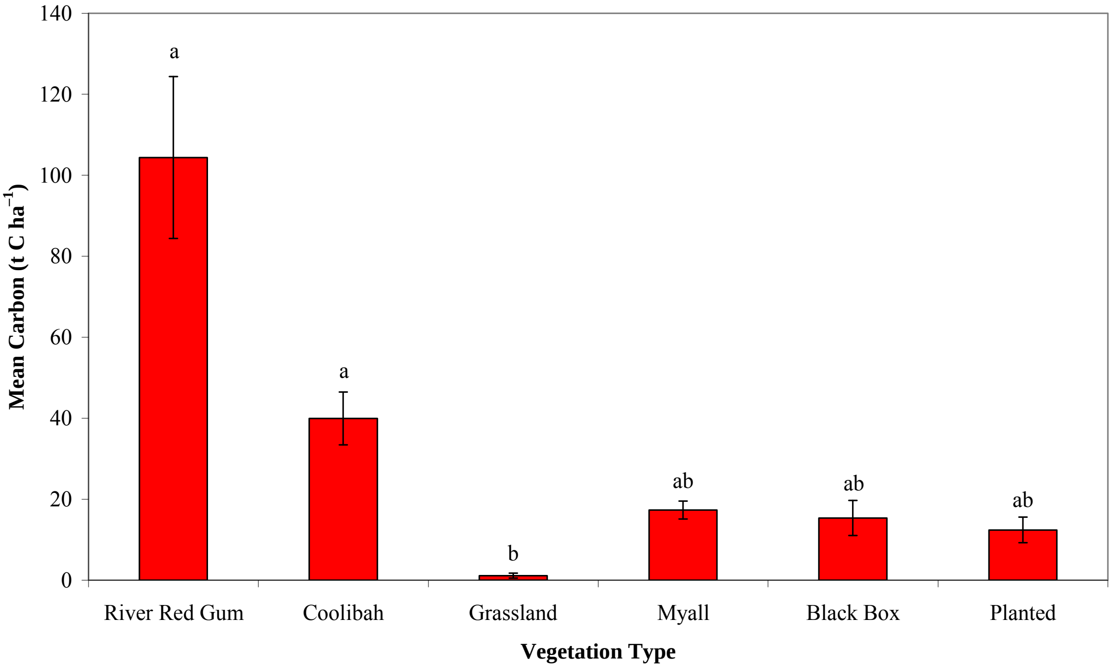


Figure 4.3: Carbon stored in above-ground woody vegetation for six vegetation types (F = 33.1, df = 5, P < 0.001). Letters represent significant differences (LSD, P < 0.05).

4.3.2 Above-ground Herbaceous Vegetation

Grasslands and planted vegetation sites had, on average, the highest levels of C in above-ground herbaceous biomass (averaged across the two sampling periods in 2008), although there was great variation among planted sites (Figure 4.4). River red gum, coolibah, myall and black box communities had significantly lower mean C contents in above-ground herbaceous biomass than grasslands. The grassland group of sites had a mean of 1.0 t C ha^{-1}, the planted group had a mean of 1.1 t C ha^{-1} and the river red gum

– coolibah – myall – black box group had a mean of 0.7 t C ha^{-1}. The largest quantity of C in herbaceous vegetation (2.0 t C ha^{-1}) was recorded in a grassland north of Pilliga.

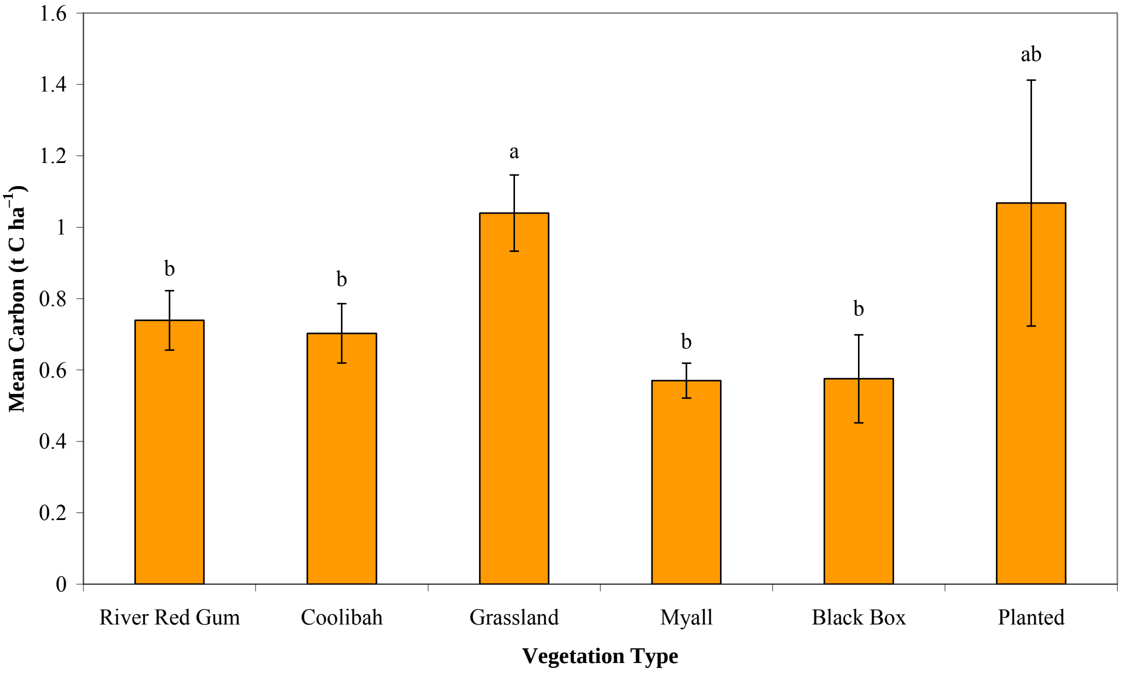


Figure 4.4: Carbon stored in above-ground herbaceous vegetation for six vegetation types (F = 2.54, df = 5, P = 0.039). Letters represent significant differences (LSD, P < 0.05).

4.3.3 Litter

Planted and river red gum sites produced the largest amount of C in litter biomass (averaged over two sampling periods in 2008) of the six vegetation types, with a mean of 2.3 t C ha^{-1} (Figure 4.5). Variability in litter accumulation was high in planted sites. The C stored in litter in grassland sites was significantly lower with a mean 0.7 t C ha^{-1}, and intermediate levels were recorded in coolibah, myall and black box-dominated vegetation with a mean of 0.9 t C ha^{-1}. The largest quantity of C in litter (4.2 t C ha^{-1}) was recorded in a mixed tree and shrub planting west of Narrabri.

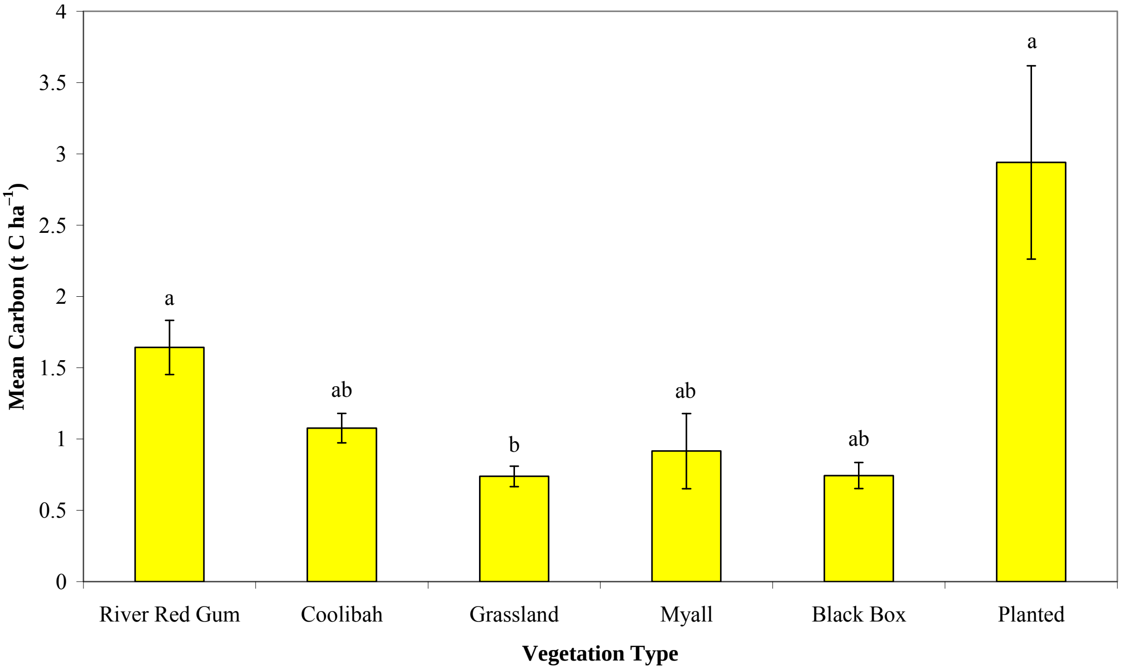


Figure 4.5: The C stored in litter in six vegetation types ($F = 7.65$, $df = 5$, $P < 0.001$). Letters represent significant differences (LSD, $P < 0.05$).

4.3.4 Dead Standing Wood

River red gum sites had the largest quantities of C in dead standing wood (average of 4.7 t C ha^{-1}), while grassland and planted sites had negligible quantities (Figure 4.6) (average of 0.01 t C ha^{-1}). The coolibah group had an average of 1.8 t C ha^{-1} in dead standing timber and the myall – black box group had an average of 0.9 t C ha^{-1}. The largest C store in dead standing wood was 18.1 t C ha^{-1} in a river red gum-dominated site south-west of Wee Waa.

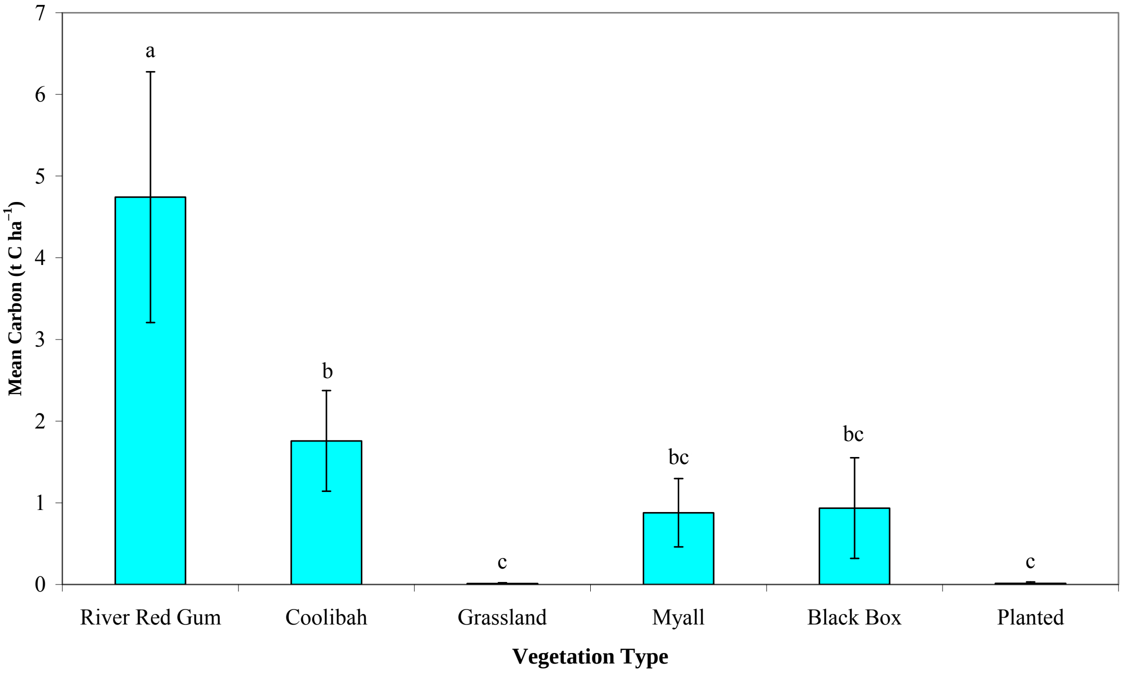


Figure 4.6: Dead standing C in six vegetation types (F = 6.05, df = 5, P < 0.001). Letters represent significant differences (LSD, P < 0.05).

4.3.5 CWD

River red gum sites had the largest quantities of C in CWD (6.2 t C ha^{-1}), although myall and coolibah-dominated sites also had significant quantities of C stored in CWD (2.4 t C ha^{-1} and 2.3 t C ha^{-1}, respectively) (Figure 4.7). Grassland and planted sites contained very little C stored in CWD (average 0.01 t C ha^{-1}), as did black box-dominated sites (0.4 t C ha^{-1}). The greatest quantity of C stored in CWD (27.3 t C ha^{-1}) was recorded in a river red gum-dominated site near Boggabri.

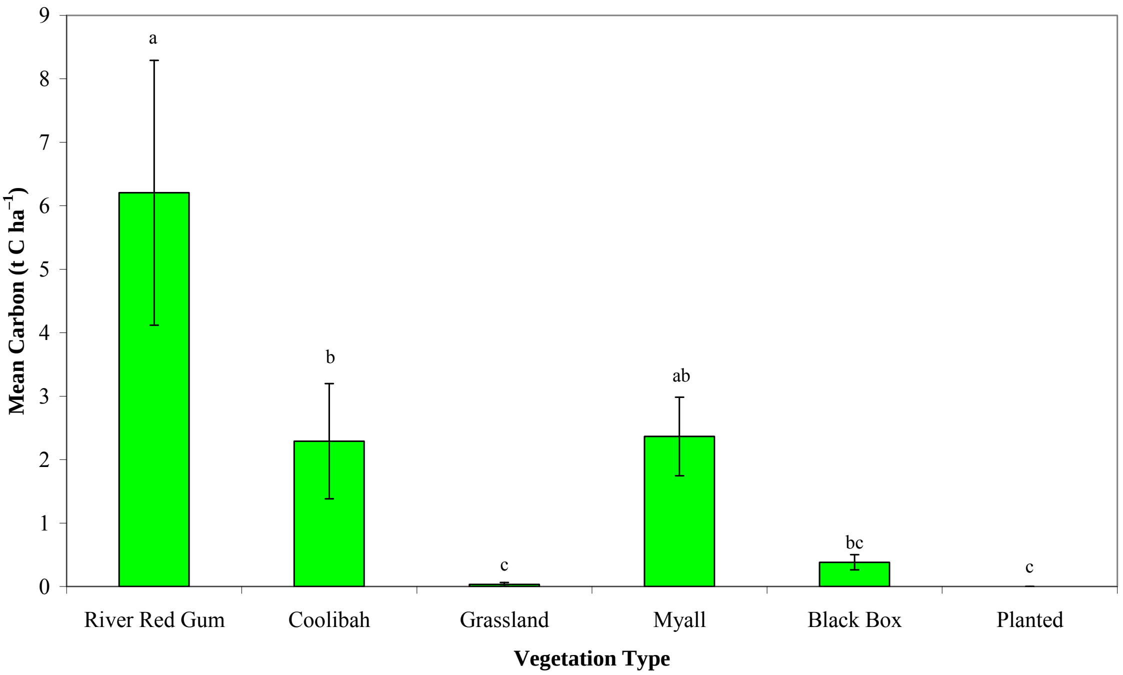


Figure 4.7: The C content of CWD in six vegetation types (F = 10.8, df = 5, P < 0.001). Letters represent significant differences (LSD, P < 0.05).

4.3.6 Above-ground Biomass

River red gum sites had the largest quantities of C stored in above-ground biomass (117.7 t C ha^{-1}). Coolibah-dominated sites also had significant quantities of above-ground biomass C (45.8 t C ha^{-1}) (Figure 4.8). Grassland sites contained very little above-ground C (2.9 t C ha^{-1}), and planted, myall and black box-dominated sites averaged 18.8 t C ha^{-1}). The greatest quantity of C stored above-ground (241.3 t C ha^{-1}) was recorded in a river red gum-dominated site on a stock route east of Narrabri.

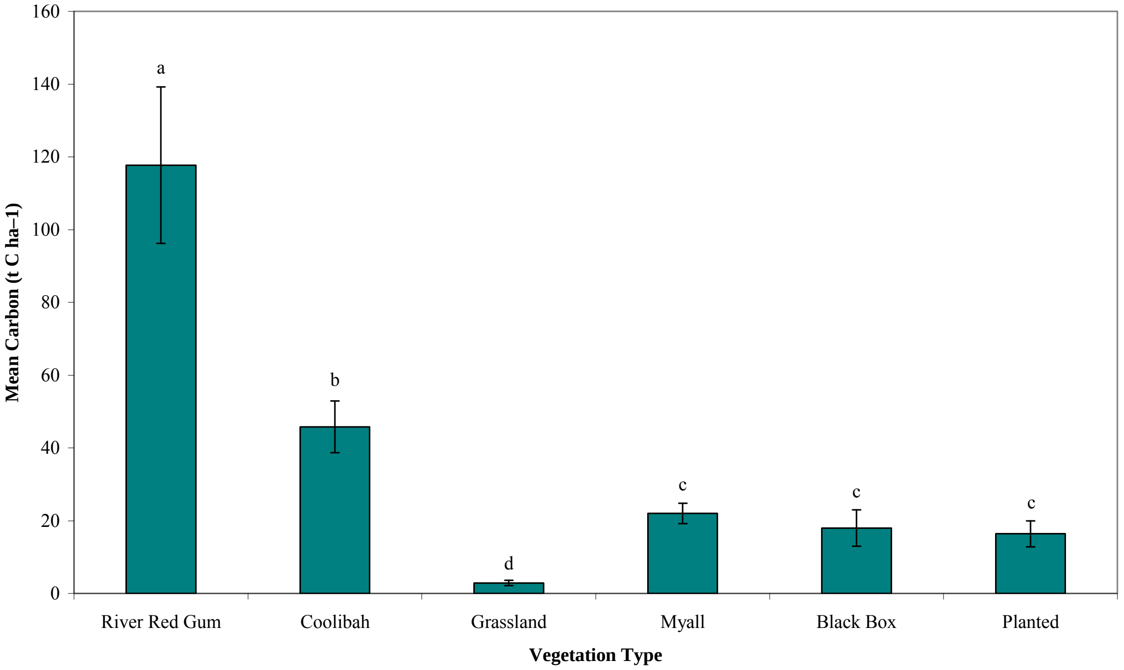


Figure 4.8: Total above-ground C for six vegetation types (F = 32.2, df = 5, P < 0.001). Letters represent significant differences (LSD, P < 0.05).

4.3.7 Roots

Because estimates of root biomass were made from above-ground woody and herbaceous biomass, the pattern of root C was similar to the pattern of above-ground biomass C. Root biomass, and therefore root C, was greatest in river red gum sites (average 26.5 t C ha^{-1}) (Figure 4.9). Root C was lowest in grassland sites at 0.8 t C ha^{-1}. In coolibah sites, root C was estimated to be 10.3 t C ha^{-1} and in the myall – black box – planted group, average root C was 4.1 t C ha^{-1}. The greatest root C store was estimated to be 59.55 t C ha^{-1} in stock route site east of Narrabri.

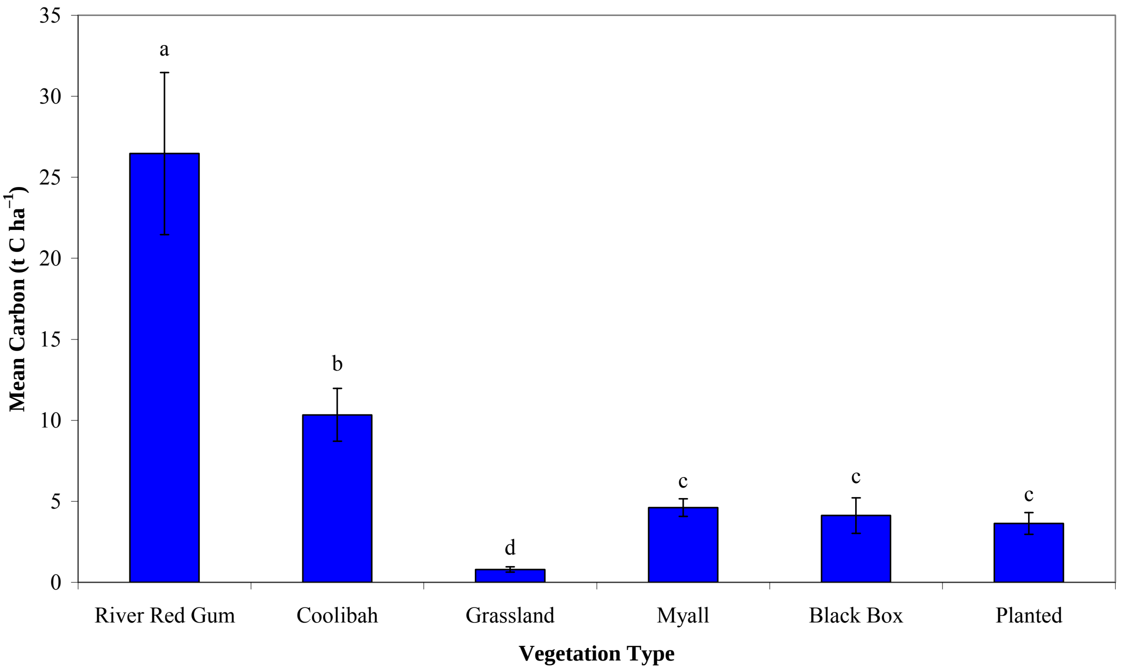


Figure 4.9: Root C for six vegetation types (F = 27.9, df = 5, P < 0.001). Letters represent significant differences (LSD, P < 0.05).

4.3.8 Soil

River red gum soils had a significantly higher average TOC content in the surface 0–30 cm of the soil profile than other vegetation types (Figure 4.10), which contained similar TOC quantities (average 39.4 t C ha^{-1}). Site values ranged from 16.8 t C ha^{-1} TOC in a grassland site near Walgett, to 98.6 t C ha^{-1} TOC in a river red gum site near Pilliga. Organic C in river red gum soils averaged 71.7 t C ha^{-1} (Figure 10). Interestingly, while not significantly different, black box soils contained less TOC (29.694 t C ha^{-1}) than grasslands (36.4 t C ha^{-1}) whereas all other vegetation types contained more TOC than grasslands (Figure 10).

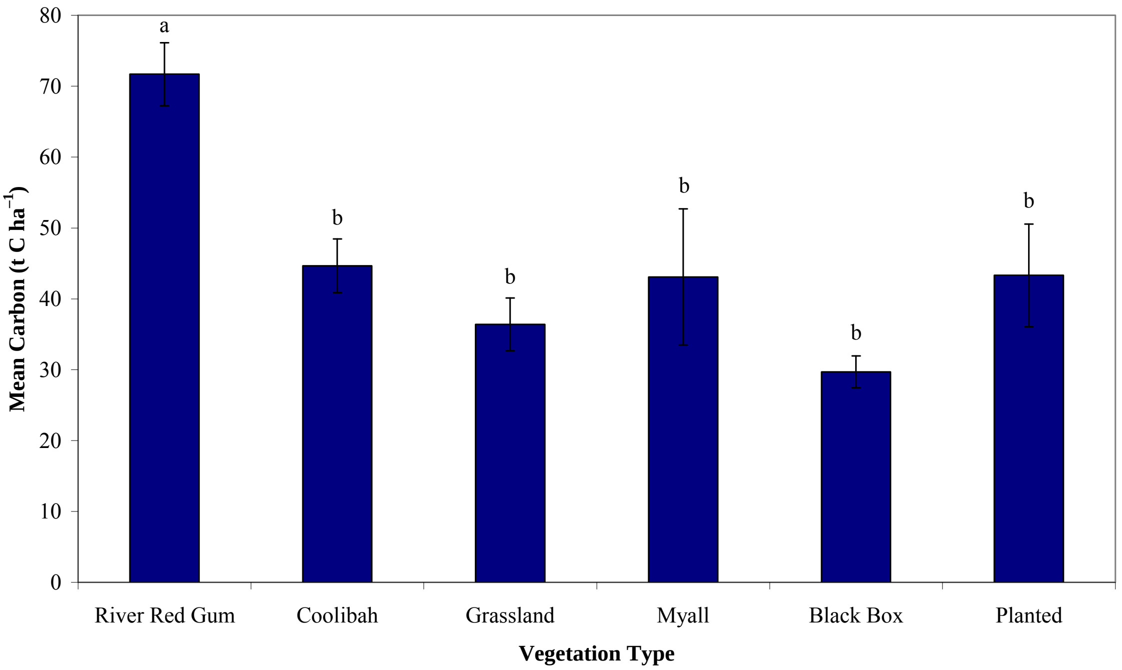


Figure 4.10: Soil organic C in the top 30 cm for six vegetation types (F = 8.04, df = 5, P < 0.001). Letters represent significant differences (LSD, P < 0.05).

River red gum soils consistently had significantly higher TOC down the profile to 30 cm depth (F = 8.04, P < 0.001) compared to the other vegetation types (Figure 4.11). The distribution of C in the profile differed with vegetation type: river red gum and coolibah-dominated vegetation showed a much sharper decrease in TOC content between the 0 – 5 cm and 5 – 10 cm depth intervals than the other vegetation types, which had largely similar TOC distributions with depth.

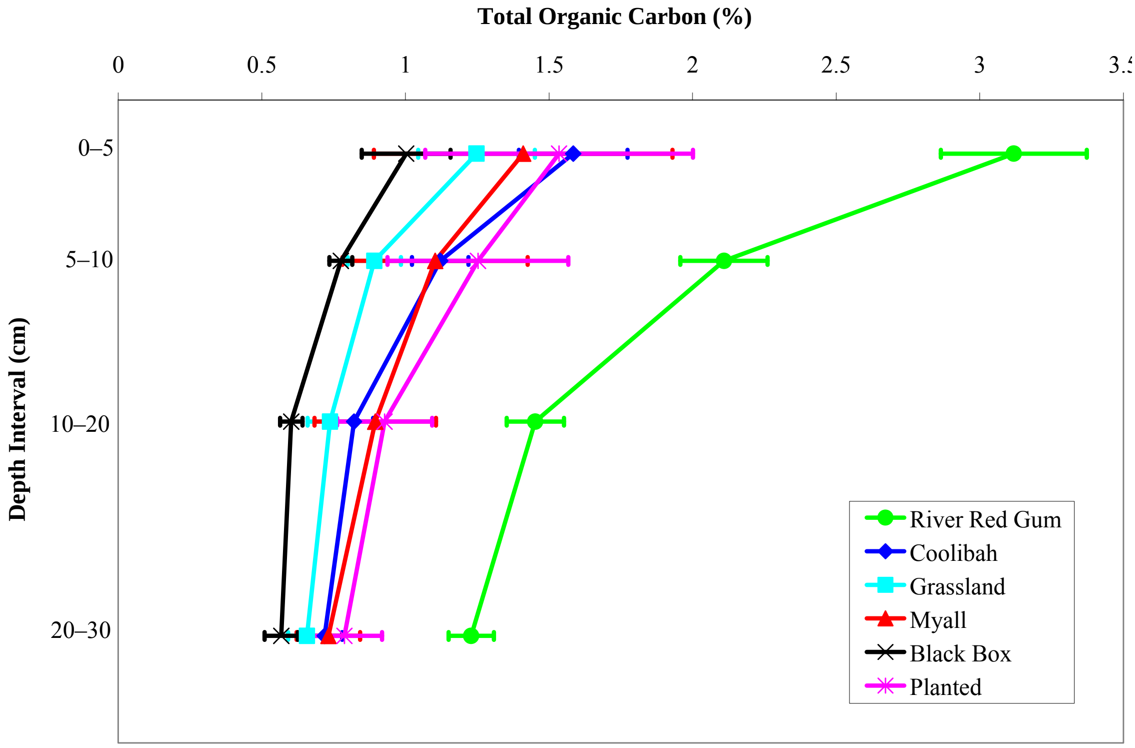


Figure 4.11: Mean TOC distribution by vegetation type ± 1 S.E. River red gum sites, on average, consistently had significantly higher %TOC concentrations throughout the soil profile when compared to other vegetation types ($P < 0.05$).

4.3.9 Total Site C

The largest quantities of C were stored in river red gum-dominated vegetation (Figure 4.12). Grassland vegetation contained the least C (40.1 t C ha^{-1}). The largest total C store (396.4 t C ha^{-1}) was estimated for a river red gum site on a stock route east of Narrabri.

In general, the largest proportion of total site C in each of the communities was contained in the upper 30 cm of the soil, although woody vegetation also contributed large quantities of C to the overall store. Only minor amounts of C were found in herbaceous vegetation, litter, CWD, dead standing woody vegetation and plant roots, these components together averaging 7–18% across vegetation types (Figure 4.13).

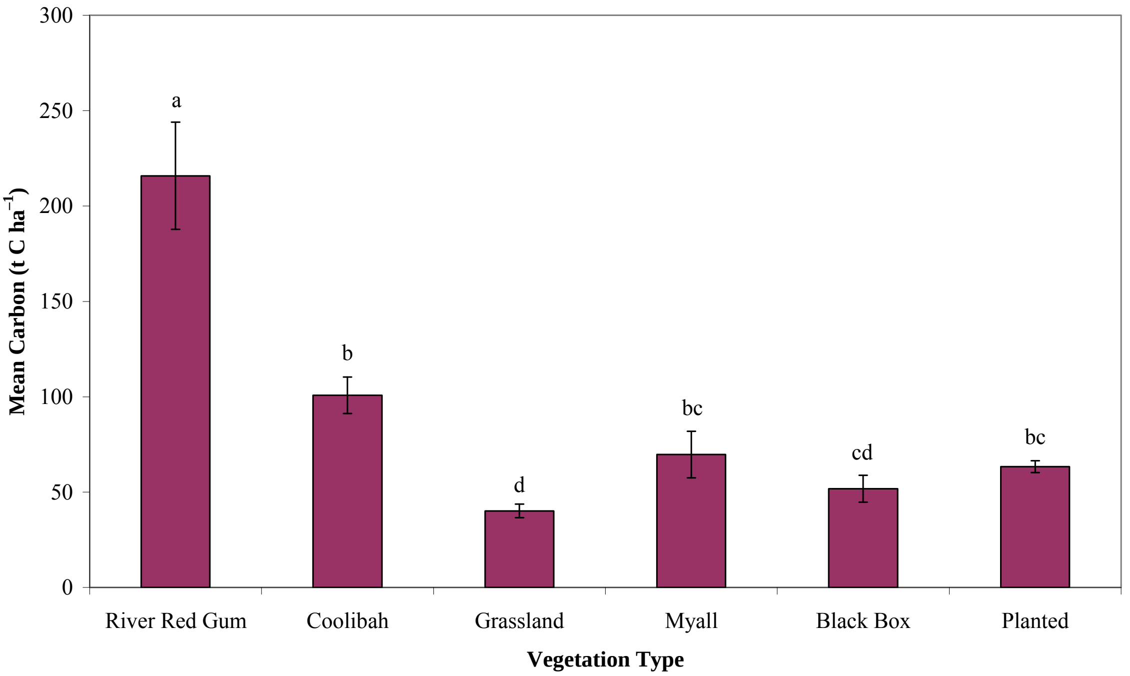


Figure 4.12: Mean total site C by vegetation type (F = 26.7, df = 5, p < 0.001). Letters indicate significant differences (LSD, P < 0.05).

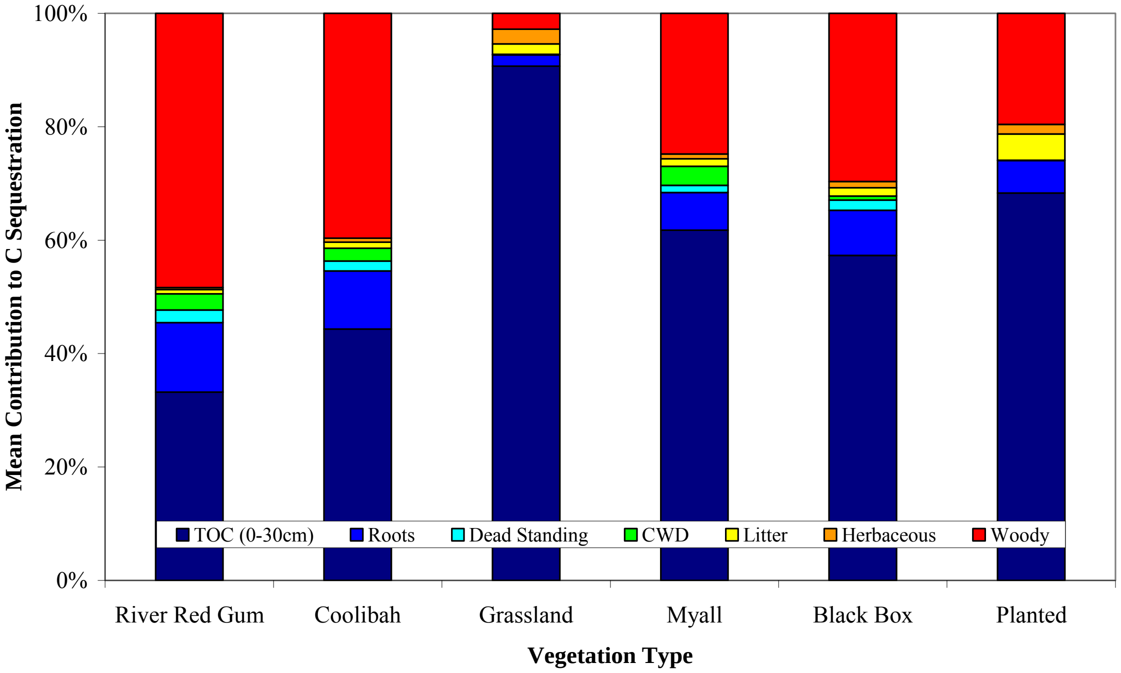


Figure 4.13: Proportion of mean total site C contributed by each ecosystem component.

4.3.10 Vegetation Influences

Spearman correlations showed significant relationships between above-ground woody vegetation C and the C contribution of all other ecosystem components (Table 4.2). The C stored in soil, litter, CWD and dead standing woody components was positively correlated with above-ground woody C, while above-ground herbaceous C was negatively correlated with above-ground woody C. Soil C was highly correlated with litter C ($r_s = 0.391$, $P < 0.001$) and significantly related to CWD C ($r_s = 0.321$, $P = 0.012$). Above-ground herbaceous C was negatively correlated with CWD C ($r_s = -0.388$, $P < 0.001$) and dead standing wood C ($r_s = -0.393$, $P < 0.001$). The C contributions of dead standing wood and CWD were positively correlated ($r_s = 0.692$, $P < 0.001$).

Table 4.2: Spearman correlation matrix for C contribution in different ecosystem components (n = 61).

	Woody	Herbaceous	Litter	CWD	Dead standing	Root	$TOC_{0–30}$
Herbaceous	–0.26*						
Litter	0.55**	–0.23					
CWD	0.80**	–0.39**	0.38**				
Dead standing	0.66**	–0.39**	0.36**	0.69**			
Root	0.98**	–0.17	0.55**	0.79**	0.64**		
$TOC_{0–30}$	0.46**	–0.03	0.39**	0.32*	0.18	0.44**	
Total	0.89**	–0.20	0.59**	0.72**	0.57**	0.88**	0.77**

*$P \leq 0.05$

**$P \leq 0.01$

4.4 Discussion

Results from this study illustrate the value of river red gum-dominated ecosystems in storing large amounts of C. Above-ground woody vegetation was a strong determinant of total site C sequestration as it strongly influenced all other C storing components of sites. Soil C storage was appreciable in many circumstances, but again this was largely determined by organic matter inputs from woody vegetation. The significant positive relationship between above-ground woody vegetation C and soil C indicates that woody vegetation has a marked influence on soil C (Table 1).

The largest total site C across the 62 sites examined in this study was 396.4 t C ha^{-1}, recorded in a river red gum-dominated woodland containing trees in excess of 2 m DBH and 95 t C ha^{-1} in the top 0–30 cm of the soil. Eight sites included in this study had C stores in excess of 200 t C ha^{-1}, all being river red gum sites. The lowest recorded C store, 21.9 t C ha^{-1}, was a derived grassland site near Walgett in the far west of the study region. Of all sites examined in this study, the 12 lowest total site C values were grasslands (of 17 grasslands studied, or 70%). This emphasises the value of woody vegetation in determining vegetation condition in terms of C sequestration, as suggested by Wilson et al. (2008) and Young et al. (2005).

4.4.1 Above-ground Woody Vegetation

River red gum and coolibah-dominated vegetation had the largest amounts of above-ground woody biomass C. In this semi-arid region, the largest quantities of woody vegetation and total site C were found close to the major water sources of the Namoi River and its anabranches. Both surface and subsurface water are less limiting along major watercourses compared to the surrounding floodplain, and nutrients are likely to be deposited in riparian areas by successive minor floods. Riparian areas are renowned for having greater NPP than other parts of the landscape (Naiman et al. 2005), due to the concentration of moisture and nutrients.

River red gum trees have rapid growth rates and often attain great size compared to other woody species in semi-arid areas (Cunningham et al. 1992). Isotope studies have shown that river red gum trees draw water from both surface and subsurface sources depending on availability, thereby avoiding water stress (Bacon et al. 1993; Mensforth et al. 1994; Thorburn et al. 1994; Thorburn & Walker 1994). Subsurface water is the favoured water supply for river red gums and is important for long-term maintenance and survival, while surface floods increase growth rates and leaf size episodically (Bacon et al. 1993). Sites where trees were stressed contained less woody vegetation biomass as river red gum trees were smaller on average, or were less dense owing to tree death (Figure 4.14).

Figure 4.14: (a) A river red gum site largely unaffected by dieback and containing the greatest woody vegetation biomass of the sites surveyed. (b) An intermediate site in the dieback – woody biomass continuum: the trees at this site are severely stressed but most are still alive. (c) A site severely affected by dieback: many of the trees in this site are dead and the least woody biomass of river red gum sites was found at this site.

Coolibah trees also reach considerable size under favourable conditions, though this species probably has slower growth rates as indicated by the density of the wood (up to 1190 kg m^{-3} for coolibah compared to maximum of 975 kg m^{-3} for river red gum; Boland et al. 2006). Coolibah generally occurs on smaller watercourses but is inundated during periods of significant overland flow. Little research has been conducted into the use of groundwater by coolibah, though use of groundwater is probable owing to the size of old growth trees in the prevailing climate, and the use of groundwater by the closely related black box (Thorburn et al. 1993) and poplar box (Banks 2006).

Black box and myall-dominated vegetation contained less woody vegetation biomass than river red gum and coolibah, and therefore less C. Black box vegetation on the lower Namoi floodplain is near the northern extent of its distribution, and much of this vegetation type occurred as stunted, multiple-stemmed trees, mostly the result of ringbarking and subsequent coppicing. Black box-dominated woodlands occur away from the main river channels on brown clay soils in the western (drier) part of the study region. Individuals of this species on the lower Namoi floodplain rarely attain dimensions reported for the species on the southern NSW and Victorian floodplains. Myall occurs across the study region, predominantly as tall shrublands. Individuals generally branch below 1.3 m and rarely reach the size of coolibah or black box. The largest individual myall tree had a DBH of 57 cm (single stem at 1.3 m) but the majority of individuals had DBH < 30 cm.

4.4.2 Limitations of Allometric Equations in Estimating Tree Biomass

Allometric equations based on relationships between DBH or height, and total tree biomass, developed by the Australian Greenhouse Office (AGO), were used in this study. Specific equations were not available for many of the species encountered so general equations representing a wide range of species in similar vegetation types had to be used. Use of these general equations is likely to have introduced errors into the data. For coolibah, a separate study indicated that the AGO equation underestimated total tree biomass (R. Smith unpub. data). Allometric equations for trees in remnant vegetation on the Namoi floodplain have otherwise not been developed, and very few allometrics have been developed in similar climatic conditions, vegetation or on similar soil types outside the Namoi floodplain. The range of tree sizes used in development of general allometric equations was smaller than the range of tree sizes encountered in this study and the applicability of the AGO equations to very large trees (such as some of the river red gums in this study) needs to be verified.

The current AGO allometric equations are based on relationships with DBH only for trees and height only for shrubs. The largest source of error in the results is likely to be due to differences in wood density. For example, the wood density of river red gum ranges between 735 kg m^{-3} and 975 kg m^{-3} while coolibah wood density ranges between 980 kg m^{-3} and 1190 kg m^{-3} (Boland et al. 2006), yet the same allometric equation had to be applied to both species. Incorporation of wood density would increase the accuracy of the AGO equations.

4.4.3 Above-ground Herbaceous Vegetation

Above-ground herbaceous vegetation contributed very little to total site C storage, even in grassland where it contributed an average of only 2.6% of total site C. Herbaceous C was in largest quantities in planted and grassland sites, and open woodland and open shrubland where competition from trees and shrubs was minimal. The variation in herbaceous production across planted sites was, in part, due to the degree of competition from planted species. Older plantings, where canopy cover was greater, had lower above-ground herbaceous biomass compared to more open, younger sites. Unlike the other vegetation types sampled, no planted sites were grazed, and some were mown regularly during early establishment (R. Watson, pers. comm., 2005).

4.4.4 Litter

Litter contributed 1–5% of total site C and was in largest quantity in environmental and agroforestry plantings, and river red gum-dominated sites, although litter production in planted sites was highly variable. Litter mass was positively correlated with woody vegetation biomass and all those factors associated with it (CWD, dead standing trees, TOC and root biomass), but negatively correlated with above-ground herbaceous biomass. Litter was negatively correlated with above-ground herbaceous vegetation C because the greatest contribution to litter was from woody vegetation and increasing herbaceous biomass was associated with decreasing woody biomass.

The largest litter C, 4.2 t C h^{-1}, occurred at a mixed *Eucalyptus, Corymbia, Angophora, Acacia, Melaleuca* and *Callistemon* environmental planting on a sodic grey clay soil west of Narrabri, planted in 1997 (the oldest of the plantings included in this study). Canopy cover at this site was approximately 30% and the highest of the planted sites studied, despite having a similar planting density to the other planted sites. Greater proportions of woody material lead to greater C:N ratios and therefore lower decomposition rates (Snowdon et al. 2005). None of the tree plantings examined in this study had recently been disturbed by management, hence litter accumulation is inevitable with increasing stand age. None of the planted sites had been grazed since establishment, but sites with lower litter accumulation, were regularly mown until approximately 2005 (R. Watson, pers. comm., 2006).

River red gum was the only woody species in this study that sheds bark in large quantities (Harden 2002), and was therefore capable of contributing large amounts of bark to litter. Coolibah, black box and myall trees all have persistent, box-type bark which is usually retained or shed sparingly (Boland et al. 2006), hence bark contributions to the litter fraction in these ecosystems is generally small. Along with the greater tree density and higher biomass production of river red gum sites compared to other vegetation types, the contribution of shed bark explains the larger quantities of litter recorded at these sites. From observation, it seemed that most of the litter in river red gum-dominated sites was produced in situ, not transported by floodwaters. However, some of the litter must have been deposited by floodwaters as there was often litter caught on nearby fences at the high water mark.

Small amounts of litter C in grassland sites was probably due to a combination of factors, including faster decomposition rate as a result of the low C:N ratio of the litter (Swift et al. 1979) as well as lower litter inputs. High soil C:N ratios were associated with high tree canopy cover (see Chapter 5), suggesting lower organic matter decomposition in densely wooded sites, and faster decomposition in grasslands. Most grassland sites were also grazed, either by livestock or macropods or a combination of the two, leading to reduced litter mass (Molinar et al. 2001).

4.4.5 CWD and Standing Dead Wood

Carbon storage in dead standing wood was largest in river red gum-dominated sites (average 5%), contributing about 3% of total site C in this vegetation type. The largest C storage in dead standing wood (18.1 t C ha^{-1}) was recorded in a river red gum site south-west of Wee Waa where several large river red gum trees had died, some of which had been rung. Sites affected by dieback also had a large amount of dead standing wood. One other site, a river red gum-dominated woodland near Boggabri in the east of the study region appeared to be undergoing a natural thinning process with 36% (272 of 752) standing stems dead. Standing dead wood C was positively correlated with the C in above-ground woody vegetation, litter and CWD, and negatively correlated with above-ground herbaceous C.

The C in CWD was greatest in river red gum sites, averaging 6.2 t C ha^{-1}, but contributed the greatest proportion (3.4%) of total site C in myall ecosystems. The largest amount of CWD C was recorded in a river red gum-dominated site near Boggabri where 27 t C ha^{-1} was measured. CWD Cwas positively correlated with the C in above-ground woody biomass, dead standing wood, and the soil, and negatively correlated with above-ground herbaceous C. The latter was probably due to the significant negative correlation between herbaceous C and woody vegetation C.

River red gum trees are renowned for shedding very large branches (Cunningham et al. 1992). Many of the river red gum sites included in this study contained very large, old trees and some of the CWD in these sites had diameters in excess of 1 m. The site with the largest quantity of CWD showed evidence of thinning, felled stems remaining on the ground as CWD. This site also appeared to be undergoing natural thinning, with 36% of stems (272 of 752) dead. The number of stems at this site was much higher than any other site.

Evidence of firewood collection was recorded at many of the river red gum sites, especially those located on public land (i.e. travelling stock routes). Firewood collection and thinning activities accounted for the large variability in CWD in river red gum sites.

The large proportionate contribution of CWD to the C store in myall-dominated sites was explained by the fact that myall is favoured for stock feed in drought times (Beadle 1981), and much of the CWD in myall-dominated sites had been cut for this purpose. The growth habit (angle of branching) and the weight of the canopy in myall trees also appeared to encourage the production of CWD. Many of the large myall trees were split at the lowest fork in the trunk, creating large fallen debris (Figure 4.15).

Figure 4.15: The branching habit of myall trees leads to production of large CWD as the canopy becomes too heavy, causing the trunk to split at the lowest fork.

4.4.6 Soil and Roots

Average organic C content (TOC) in the surface 0–30 cm of the soil was largest in river red gum-dominated sites (71.7 t C ha^{-1}) and lowest in black box-dominated vegetation (29.7 t C ha^{-1}). However, with the exception of river red gum soils, the TOC content did not differ significantly among vegetation types. Site values for soil TOC content ranged from 16.8 t C ha^{-1} to 98.6 t C ha^{-1}. The site with the largest soil TOC was a river red gum site on the edge of the Namoi River near Pilliga. Soil TOC was positively correlated with the c content of the woody vegetation, litter, dead standing material, CWD and roots. These variables were all positively correlated with woody vegetation C, suggesting that above-ground woody biomass is the main contributor to TOC, and that TOC should be interpreted in conjunction with measurements of above-ground woody biomass. Root biomass was calculated from above-ground data and therefore mimicked the woody vegetation biomass data.

River red gum soils showed significantly higher TOC throughout the upper 30 cm of the soil profile than other vegetation types. The rate of change in TOC with depth down the profile differed in river red gum and, to a lesser extent, coolibah soils compared to other vegetation types. In the other vegetation types, there was only a slight decrease in TOC at each depth, but the change was much more pronounced in river red gum and coolibah sites, especially between the 0–5 cm and 5–10 cm intervals. The concentration of SOC at the surface indicated that the soils in river red gum sites, and perhaps also the coolibah sites, received greater C inputs at the surface as opposed to deeper in the soil profile. This result is consistent with data presented by Jobbágy and Jackson (2000) who reported that the majority of soil C in forested ecosystems is found in the surface soil, more so than in grasslands. This was due to the way different vegetation types partitioned biomass, in grassland ecosystems a high proportion of annual production occurs below ground and this is the greatest input of C to the soil, whereas in forested ecosystems the dominant C input from the vegetation to the soil is through litter deposition.

On the lower Namoi floodplain, river red gum vegetation occurs as ribbons along major water courses and is predominantly confined to the edges of the Namoi River channel. East of Wee Waa, the vegetation type extends away from the primary river channel into regularly inundated floodplain depressions dissected by small channels. These areas are highly productive and, as with riparian vegetation elsewhere (Naiman et al. 2005), produce large amounts of biomass quickly in response to floods. Between floods, NPP in riparian vegetation is generally greater than that of the surrounding vegetation (Naiman et al. 2005). High NPP translates into greater litter inputs to soil and greater TOC concentration in riparian soils.

4.4.7 Influence of Vegetation Condition on Total Site C

This study illustrated the importance of woody vegetation, especially large, old trees, to C storage, both in terms of the woody component itself, and for the influence of woody vegetation on soil C stores. A single tree with a DBH of 100 cm stores the same amount of C as 875 saplings with a DBH of 5 cm. This suggests that, to achieve maximum C storage, management should protect large, old trees and encourage smaller trees to attain greater size. Dieback across the lower Namoi floodplain (e.g. Reid et al. 2007) reduces the ability of river red gum-dominated vegetation to store C by stunting trees, slowing growth rates, and over long time periods, contributing to increased C release to the atmosphere through decomposition and respiration. Factors contributing to dieback need to be identified and management implemented to minimise its impact. Research by Reid et al. (2007) suggests that the causes of dieback are complex and likely due to several factors in combination.

The contribution of TOC to total ecosystem C varied with vegetation type. In grassland soils, 90% of total ecosystem C storage came from the upper 30 cm of soil whereas, for river red gum soils, 30% of total C storage was contributed by the soil component. However, overall quantities of TOC were higher in river red gum soils. These results illustrate the importance of encouraging management to protect soil C reserves.

4.5 Conclusion and Future Research

Presence of woody vegetation biomass was the most important determinant of vegetation condition for C storage, both as a C sink in its own right, and in generating litter, CWD, dead standing wood, root and soil C. River red gum-dominated vegetation was of greatest value in terms of total C storage, reflecting a world-wide pattern of high NPP in riparian vegetation compared to other vegetation types in similar environments.

Given the importance of woody vegetation for C storage, future studies are required to determine the accuracy of existing allometric equations for species of the lower Namoi floodplain and to develop allometric equations in natural conditions in similar climatic conditions and soil types where errors associated with existing equations are unacceptable. In the interim, incorporation of a factor relating to wood density may lead to more accurate allometric equations.

Chapter 5: Erosion Mitigation Value of Native Vegetation of the Lower Namoi Floodplain

Erosion of a sodic grey clay (vertosol) near Pilliga after a flood (April 2008).

Chapter 5: Erosion Mitigation Value of Native Vegetation of the Lower Namoi Floodplain

5.1 Introduction

Accelerated soil erosion affects 1094 million ha of land globally and is the most widespread form of land degradation (Oldeman 1994). Soil erosion has been recorded from very early times (Neumann 1980), has caused the demise of numerous civilisations, and is still a major concern in agricultural regions on all inhabited continents (Carter & Dale 1974; Diamond 2005; Lal 1990; Lal & Stewart 1990; McNeill & Winiwarter 2004). Soil erosion reduces soil fertility and water quality, and can result in major soil loss and gully formation (Lal 1990). Remediation of such degradation can be very costly (Pimentel et al. 1995).

Vegetation provides an ecosystem service by limiting or reducing the costs associated with soil erosion (MA 2003; Costanza et al. 1997; Daily 2000; de Groot et al. 2002; Norberg 1999). Vegetation: (1) reduces raindrop impact through provision of ground cover and canopy interception (Lang 1984); (2) increases infiltration and reduces runoff (Bergkamp 1998; Selby 1993); and (3) increases aggregate stability (Cerdà 1998, Eviner & Chapin 2002). However, depending on its location, composition, structure and function, vegetation differs in its provision of, and value for, erosion mitigation (Cerdà 1998; Costanza et al. 1997).

Soil erodibility is a measure of the susceptibility of soil to detachment and transport when subjected to the applied forces of raindrop splash or overland flow, and varies with soil type (Rosewell & Loch 2002). Soil aggregate stability refers to the ability of soil structural units to resist detachment and maintain aggregates greater than clay particle size when wetted (Yates 1972). Within a soil type, erodibility is determined by aggregate stability and those factors influencing aggregate stability.

Vertosols are highly erodible (Freebairn et al. 1996; Freebairn & Wockner 1986) and Australia has a greater area (880 000 km^2) of this soil type than any other country (Ahmad 1996; McKenzie et al. 2004). Vertosol soils are distributed extensively in Queensland, New South Wales and the Northern Territory with some minor occurrences in most other states (Peverill et al. 1999). The value of these soils for irrigated and dryland cropping and grazing has resulted in large areas of the original native vegetation being cleared or modified (Ahmad 1996). Research into the structural characteristics of vertosols in agricultural situations has revealed an array of problems including low aggregate stability (Prebble 1987), low infiltration (Little et al. 1992) and high surface crusting and sealing (Pillai-McGarry & Collis-George 1990), leading to severe sheet erosion (Junor et al. 1979) and crop yield declines (Yates 1972).

Soil aggregation is determined by soil pedogenic, physical, chemical and biological processes, but the importance of these processes in determining macro-aggregate (>250 μm) and micro-aggregate (<250 μm) stability differs (Amézketa 1999; Bronick & Lal 2005; Oades 1984; Tisdall & Oades 1982). Slaking (the breakdown of soil into smaller aggregates when dry aggregates are subjected to rapid wetting) is an indicator of macro-aggregate stability. Forces contributing to aggregate stability at the macro-aggregate scale are predominantly driven by organic matter dynamics in most soils, which makes macro-aggregate stability sensitive to management (Oades 1984; Tongway & Whitford 2002). Dispersion of clays (disintegration of aggregates into individual components of sand, silt and clay) is an indicator of micro-aggregate stability, and is determined by soil chemistry, although organic matter also plays a role at this scale. In vertosols, as in most other soils, multivalent cations such as calcium are important in increasing micro-aggregate stability, while sodium promotes clay dispersion (Sumner 1993).

Soils with high aggregate stability do not disperse, slake very little or not at all, and the clay particles tend to flocculate if disaggregated when subjected to immersion wetting (Rengasamy et al. 1984). Dispersion and severe slaking can cause a major loss of soil structure, causing problems such as poor aeration, surface crusting, cloddiness, low water intake, poor drainage and waterlogging (Geeves et al. 2000). Dispersion and slaking are therefore dominant causes of soil erosion. A degree of slaking of aggregates has been experienced in the majority of studies examining aggregate stability in vertosols, and is now almost accepted as an inherent property of the soil type (Cass 1999; Coughlan 1984). Mechanisms for aggregate stabilisation in vertosols have been attributed to physico-chemical characteristics including clay mineral type (Freebairn et al. 1996; Dalal 1989; Dalal & Bridge 1996; Prebble 1987; Yates 1972; Yates & McGarity 1984). The influence of organic matter in vertosols has been unclear (Coughlan & Loch 1984). However, a decline in structure as a result of a decline in organic matter has been reported (Dalal & Chan 2001; Hubble 1984). Most observations regarding the structural characteristics of vertosols have been made in cropping situations (see Dalal & Chan 2001 for examples).

Studies have tracked the decline in organic carbon and aggregate stability after conversion of native vegetation to cropland (e.g. Chan et al. 1988, 1995; Cook et al. 1992; Loch 1994; Prebble 1987), and the improvement in structural stability with conversion of previously cultivated lands to pasture or grasslands (e.g. Chan et al 1997) in vertosol soils. However, few studies have examined aggregate stability and factors contributing to aggregate stability in vertosol soils in their original woodland or grassland situations, or the value of different non-crop vegetation types for provision of erosion mitigation services. In addition, little is known of the functional attributes that determine the influence of vegetation on aggregate stability in vertosols.

Raindrop impact was not a concern on the majority of the soils of the lower Namoi floodplain owing to their strongly self-mulching character. Aggregate stability was a greater concern as the soils have a high propensity to slake and disperse, allowing them to be easily eroded by floodwaters. As such, ground cover was considered explicitly in this chapter, except where it influenced aggregate stability. Cover is used in addition to aggregate stability in Chapter 7 to rank sites according to erosion mitigation value. However, this chapter concentrates on the physico-chemical characteristics of the soils.

The aim of this study was to investigate the influence of vegetation type and functioning on aggregate stability in vertosol soils. Six vegetation types occurring on vertosols and representing a large range of landscape types on the lower Namoi floodplain were included in the study. The objective was to determine the influence of these vegetation types on soil slaking and dispersion, and hence, provision of an erosion mitigation service.

5.2 Methods

5.2.1 Study Region

The lower Namoi floodplain in northern NSW, Australia (Figure 5.1), is a highly modified agricultural environment dominated by cracking clay soils prone to severe erosion from high-intensity storms when ground cover is sparse, conditions often experienced in the area (Towler 1984). The region's soils have been described and mapped by many authors including Prescott (1944), McGarity (1949), Stephens (1961), Northcote (1966), Stace et al. (1968) and Stannard and Kelly (1977). Dominant soils of the region are black, grey and brown clays typically with self-mulching characteristics, the majority with slightly alkaline to strongly alkaline pH, often increasing with depth. Texture does not change significantly with depth and is characterised by high clay content (30–65%). Many of the soils contain free lime or gypsum at varying depths. Soils throughout most of the region are classified as Vertosols in the Australian Soil Classification system (Isbell 1996).

This study was undertaken on cotton farms and travelling stock routes of the lower Namoi floodplain between Boggabri (–30.7152˚, 150.0346˚) and Walgett (–30.0318˚, 148.1012˚) (Figure 5.1). The climate of much of the region is arid and hot, grading into temperate with no dry season and hot summers in the east (Peel et al. 2007). Mean annual rainfall decreases from east to west, from 400 mm near Walgett to 600 mm near Boggabri, with a slight summer dominance, although rainfall is unpredictable in time and amount (Stannard & Kelly 1977). Rainfall events often occur as intense thunderstorms (Kearle et al. 2002; SPCC 1980), and runoff during these events can be substantial. Areas in the western portion of the catchment can experience months of little or no rainfall, making these areas susceptible to erosion as a result of a lack of ground cover, particularly when drought-breaking rain is in the form of intense storms (Kearle et al. 2002; Ward 1999).

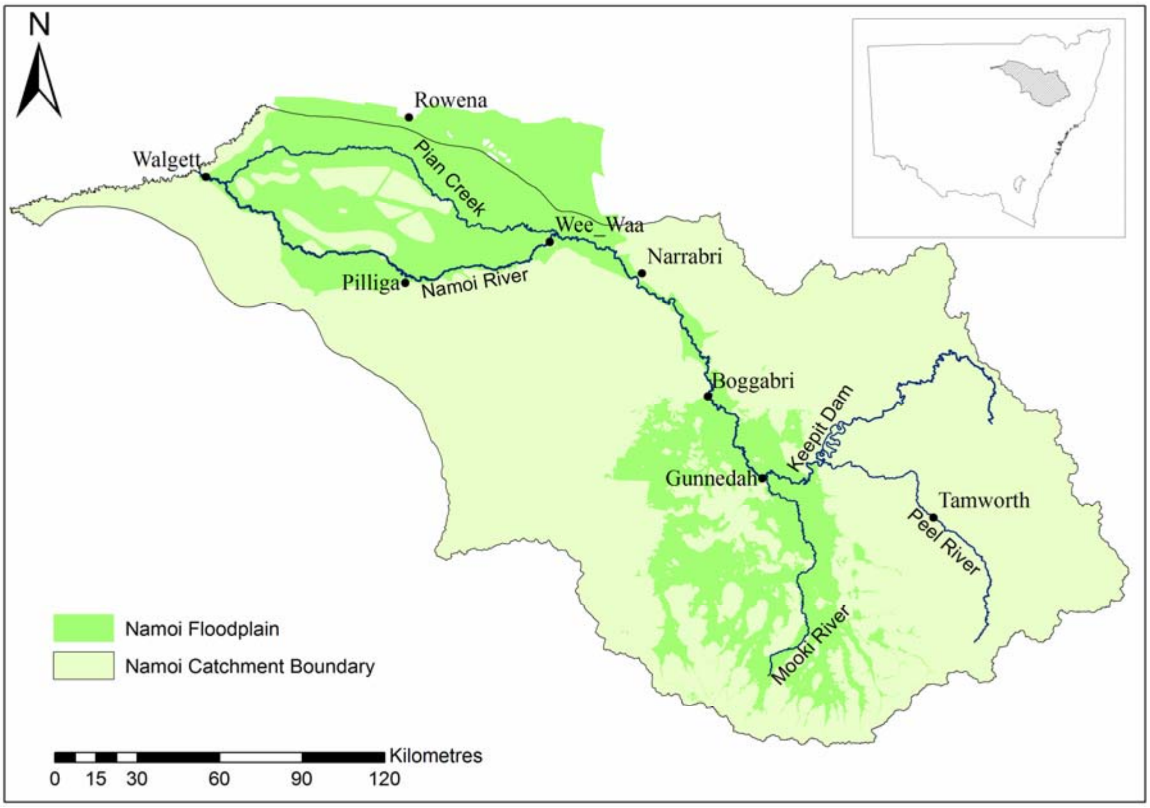


Figure 5.1: Location of the lower Namoi floodplain study region north-west of Boggabri in northern NSW.

The floodplain west of Narrabri is flat, sloping to the north-west with an average gradient of 1:2000, and becoming flatter near Walgett where the Namoi River joins the Barwon–Darling River system (Stannard & Kelly 1977). The lower Namoi floodplain was shaped by extensive low energy movement of water across the landscape and concomitant clay deposition (Kearle et al. 2002). Water movement, flood frequency, duration and inundation patterns have been altered by the extensive development of irrigated agriculture since the 1980s (SPCC 1989; Water Resources Commission 1980).

Most flood events now move across the area with increased energy, diverted along predefined paths by levees, waterways and other engineering works (SPCC 1980).

Natural vegetation patterns in the study region are influenced by soil characteristics, geomorphological characteristics and the frequency and duration of inundation by floodwaters (Stannard & Kelly 1977; Chapter 2). The dominant vegetation types on the vertosol soils of the region include river red gum (*Eucalyptus camaldulensis*)-dominated riparian forests and woodlands, coolibah (*Eucalyptus coolabah*) woodlands of varying density, black box (*Eucalyptus largiflorens*) woodlands and open woodlands, myall (*Acacia pendula*) tall shrubland and tall open-shrubland, and native and derived perennial grassland and chenopod communities. These five vegetation types plus tree plantings on cotton farms were the focus of this study.

5.2.2 Field Sampling

Soil samples were collected at 60 sites across the lower Namoi floodplain in December–January 2007–08. Sites were chosen to represent the range of management and condition states commonly observed across the six vegetation types on vertosol soils of the region. Because vegetation types varied greatly in extent in the region and not all vegetation types were represented in all condition states, a balanced design could not be achieved. Eventually, 13 river red gum, 17 coolibah, 17 grassland, six myall, four black box and four tree plantings were sampled. Management at individual sites ranged from undisturbed to continuously grazed by native macropods, and sheep or cattle. Some grasslands had previously been cultivated, and most of the tree and shrub plantings were previously cropping paddocks (predominantly cereal crops). No sites used in this study had been cultivated in the preceding 15 years. Multiple sites were located on each property, some in the same paddock where contrasting vegetation types existed.

Nine soil core samples were taken using a manual coring device from a 25 × 25 m quadrat stratified by cover at each site. Cover at the point where each core was collected was recorded as tree, shrub, legume, chenopod, grass, graminoid, forb, bare ground or litter. Cores were subdivided into depth increments of 0–5 cm and 20–30 cm. These depth increments were stored in zip-lock plastic bags in cool, shady conditions for up to 1 week during sampling and then at 4°C for up to 2 weeks prior to air-drying. Use of air-dry aggregates was considered acceptable for analysis of aggregate stability as prevailing field conditions during summer yield aggregates of similar moisture content (Chan et al. 1988).

Vegetation surveys were conducted to measure the projected foliage cover of all vascular plant species in 20 × 20 m quadrats (herbaceous vegetation) and 1 ha quadrats (shrubs and trees) at all sites. Herbaceous vegetation and litter biomass was measured using a modified BOTANAL (Tothill et al. 1978) system. Twenty 50 × 50 cm quadrats located at 4 m intervals around the perimeter of the 20 × 20 m quadrat were used to measure biomass and ground cover. Surveys were conducted in April–May and again in October–November 2008 as the weather leading up to these sampling periods was favourable for maximum vegetation growth. Hence, the vegetation survey data recorded the maximum vegetation cover and biomass potential at study sites under prevailing management. Information on land use history, stocking, cultivation, flood frequency and duration, climate and other management details were obtained from landholders.

5.2.3 Soil Analyses

Chemical analyses were used to characterise the soils of the different vegetation types and provide co-variables in analyses of the erosion mitigation value of vegetation (Table 5.1: surface and Table 5.2: subsurface). A bulked sample for each depth increment at each site was passed through a 2 mm sieve using a mechanical soil mill after large pieces of macro-organic matter such as tree roots and leaves were removed. Samples were mixed thoroughly and subsampled for further analyses. Electrical conductivity (EC) and pH were determined in a 1:5 soil-to-water extract. For soils with pH < 7.5, exchangeable cation content (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}) was determined using atomic absorption spectroscopy following extraction using 1M NH_4Cl. For soils with pH ≥ 7.5, exchangeable cations were determined at the DPI Victoria lab at Werribee using the alcoholic 1M NH_4Cl method (Tucker 1954) after pre-treatment with aqueous ethanol and aqueous glycerol to remove soluble salts. Exchangable cation values were summed to give the effective cation exchange capacity (ECEC) and the Ca:Mg ratio was calculated by dividing Ca by Mg. Exchangeable calcium (ECa), exchangeable magnesium (EMg), exchangeable potassium (EK) and exchangeable sodium (ESP) were calculated as a percentage of ECEC. Soil particle size distribution of samples was determined using the pipette method after overnight end-on-end shaking in sodium hexametaphosphate $(NaPO_3)_6$ dispersant solution (Day 1965) and expressed as proportions of sand, silt and clay using the International classification.

A subsample of each bulked soil sample was pushed by hand through a 0.5 mm sieve to remove macro-organic matter and sand particles prior to analysis of total C, total organic C (TOC) and nitrogen (N) content using a Carlo Erba NA 1500 Solid Sample

Analyser. Where sample pH > 7.5, an additional sample treated to remove carbonates using 2% orthophosphoric acid immediately followed the untreated sample. Where treatment to remove carbonates had no effect on the measured C content as determined by the difference in C content between treated and untreated samples, the average percent C value of the treated and untreated samples was reported as TOC. Where carbonates were present, the C content of the treated sample was reported as TOC. The C:N ratio was calculated by dividing % total C (%C from untreated soil samples) by the N percentage.

Water stable aggregation was measured on a subset of nine surface (0–5 cm) and 16 subsurface (20–30 cm) samples from individual soil cores representing the greatest range in soil physical and chemical variables encountered across the study region. Water stable aggregation tests the degree of slaking of large soil aggregates into smaller aggregates and is reported here as the mean weight diameter (MWD) of aggregates, higher MWD denoting greater macro-aggregate stability. Methods followed a modified version of Kemper and Rosenau (1986). A 20 g air-dry sample passing a 10 mm sieve but retained on a 2 mm sieve was placed on the top of a nest of pre-weighed sieves (2 mm, 1 mm, 0.5 mm and 0.25 mm) and gently oscillated in and out of deionised water at 15 strokes min^{-1} for 5 min. The sieves were then placed in a fan-forced oven at 105°C for 1 hour and weighed individually. Material < 0.25 mm was calculated from the difference between initial and final total sample weights. Samples from different sites were used for surface and subsurface soil analyses.

The aggregate stability in water test (ASWAT) (Field et al. 1997) rates the dispersion of soil under air-dry and plastic limit conditions and provides a relative ranking of aggregate stability against dispersion. This method was used to measure dispersion of aggregates from individual cores at two soil depths (0–5 cm and 20–30 cm) after rapid wetting. Four air-dry aggregates of approximately 5–10 mm diameter were placed in a petri dish that was slowly filled with deionised water. Scores based on the degree of dispersion (appearance of a cloud of clay particles) between 0 (no dispersion) and 4 (complete dispersion) were allocated to each petri dish after 10 min and again after 2 hours. For soils that showed no dispersion, a sample remoulded at the plastic limit was tested and scored in the same manner as the natural aggregates previously. For natural aggregates that scored ≥ 1, the score from the 10 min and 2 hour observations were summed and added to 8, giving a score ranging from 9 to 16. For natural aggregates that scored 0, the remoulded scores for the 10 min and 2 hour observations were summed giving a score from 0 to 8.

5.2.4 Statistical Analyses

5.2.4.1 Soil and Vegetation Type Characteristics

One-way AOV (on rank-transformed data where less severe transformations did not satisfy the assumption of normality for parametric AOV) and Tukey's HSD pairwise comparisons were used to investigate differences in surface and subsurface soil characteristics between vegetation types using Statistix 8 (Analytical Software 2003). The same approach was used to investigate differences in vegetation characteristics (litter biomass and cover, herbaceous vegetation biomass and cover, and canopy cover) across vegetation types.

5.2.4.2 Water Stable Aggregation

Pearson correlations were generated for all pairs of physico-chemical variables for surface ($n = 9$) and subsurface soil ($n = 16$) samples separately in Statistix 8 (Appendix 2 and 3). Variables were normally distributed or transformed to ensure normality prior to analysis and therefore satisfied the requirements for Pearson correlation analyses. Differences were reported as significant where $P \leq 0.05$, or highly significant where $P \leq 0.01$. The Schwartz (1978) Bayesian Information Criterion (BIC) in R 2.8.1 (R Development Core Team 2009) was used to find the best-fit, most parsimonious regression model to explain surface and subsurface MWD of aggregates using the backward selection procedure. Input variables for stepwise regression were chosen to avoid multi-collinearity, an issue highlighted by Pearson correlations. Thus, the starting variables in multiple regression differed for surface and subsurface soils. Variables included in the initial models were TOC, C:N, ECa, EC, Ca, Na and sand for surface samples and ln (TOC+1), C:N, K, Ca, log (EC), ECEC and EMg for subsurface soils. Regression models were fitted, and the assumption of normality of residuals was checked visually and via appropriate statistical tests (e.g. Wilk-Shapiro).

Spearman's rank correlation coefficients were used to determine relationships between vegetation (herbaceous biomass and cover, and canopy cover) and litter (biomass and cover) characteristics, and the surface and subsurface soil variables included in the final models explaining the surface and subsurface MWD of aggregates. Many of these variables were not normally distributed hence Pearson correlations could not be computed.

5.2.4.3 ASWAT

Spearman's rank order correlations in Statistix 8 determined the nature and significance of relationships between median ASWAT score (n = 9) and soil physico-chemical variables, and between all pairs of soil physico-chemical variables. Surface and subsurface samples were assessed separately. Stepwise regression using BIC was again performed in R 2.8.1 in a backwards selection procedure to find the best-fit, most parsimonious regression model to explain surface and subsurface median ASWAT scores using a combination of all soil physico-chemical variables. A suitable model could not be found for surface soils due to collinearity between predictor variables, but a linear regression model was fitted to explain subsurface soil ASWAT scores.

Principal components analysis (PCA) was performed in PRIMER 6.1.8 (Clarke & Warwick 2006) to ordinate sites in relation to all soil physico-chemical properties and median ASWAT scores for each depth. PCA reduced dimensionality in the multivariate data sets where measured variables were highly correlated, and highlighted variables that were of genuine importance in explaining ASWAT scores across sites (Manly 1994). Variables were transformed if not normally distributed and standardised to produce the correlation matrix on which PCA was performed (Clarke & Gorley 2006).

One-way AOV on rank-transformed data was used to examine the effects of ground cover on surface soil ASWAT scores from each individual core (nine cores per site, 550 cores in total). Six cover classes were investigated: trees, shrubs and legumes, grasses and graminoids, chenopods, forbs and bare ground. Tukey's pairwise comparisons were used to identify homogeneous groups across cover classes based on ASWAT score.

5.3 Results

5.3.1 Surface and Subsurface Soil Characteristics

Analysis of surface soil characteristics across vegetation types showed highly significantly differences ($P < 0.001$) for ESP, TC, TOC, N and C:N, moderately significant differences ($0.001 \leq P \leq 0.010$) for Ca, Mg, Na, ECEC, ECa, EK and silt, and significant differences ($P \leq 0.05$) for pH, Ca:Mg and EMg. No significant differences in surface soil between vegetation types were found for EC, K, clay or sand. Table 5.1 shows mean values for each surface soil variable and homogenous groups of means among vegetation types.

Table 5.1: Mean surface soil variables (0–5 cm) for each vegetation type.

	River red gum (n = 13)	Coolibah (n = 17)	Grassland (n = 17)	Myall (n = 6)	Black box (n = 4)	Planted (n = 4)
ASWAT[#]	2.00ac	5.00ab	5.00ab	5.00abc	6.50abc	6.50a
pH	7.18b	7.77ab	7.83ab	7.76ab	8.04ab	8.19a
EC (dS/m)[1]	0.11a	0.10a	0.10a	0.13a	0.12a	0.13a
Ca (cmol+/kg)	27.15a	22.80ab	20.29b	20.54b	20.18ab	18.44b
Mg (cmol+/kg)	12.79a	12.65a	10.35ab	8.00b	8.54ab	11.28ab
Na (cmol+/kg)[1]	0.37b	0.74ab	0.62ab	0.44ab	1.29ab	1.46a
K (cmol+/kg)	1.99a	2.07a	2.24a	2.66a	2.20a	2.00a
ECEC (cmol+/kg)	42.30a	38.27ab	33.51b	31.64b	32.21ab	33.17ab
Ca:Mg[3]	2.15ab	1.90b	2.09ab	2.68a	2.40ab	1.63b
ECa (%)	64.08a	59.74ab	60.56ab	64.89a	62.64ab	55.14b
EMg (%)	30.31ab	32.82a	30.51ab	25.08b	26.58ab	34.43a
ESP[1] (%)	0.87b	1.89ab	1.97a	1.38ab	3.90ab	4.15a
EK[2] (%)	4.74c	5.56bc	6.96ab	8.65a	6.89abc	6.29abc
TC (%)[1]	3.14a	1.61b	1.27b	1.47b	1.04b	1.70ab
TOC (%)[1]	3.12a	1.58b	1.25b	1.41b	1.00b	1.54ab
N (%)[1]	0.31a	0.16b	0.15b	0.17b	0.22ab	0.17ab
C:N[3]	10.22a	10.26a	8.49b	8.78ab	7.37ab*	10.86a
Clay (%)	44.26a	48.87a	44.59a	39.78a	36.78a	43.18a
Silt (%)	26.00ab	18.79c	20.27bc	22.29abc	23.37abc	30.82a
Sand (%)[2]	29.75a	32.34a	35.14a	37.93a	39.86a	26.00a

[1] rank-transformed; [2] Ln-transformed (ln(X+1) where applicable); [3] log-transformed (Log(X+1) where applicable) to attain normality prior to AOV; [#] median values; *single outlier excluded. Letters indicate significant differences (Tukey's) where $P \leq 0.05$.

Analysis of subsurface soil characteristics showed highly significantly differences ($P < 0.001$) between vegetation types for pH, EC, Ca, Na, Ca:Mg, ECa, EMg, ESP, TC, TOC and N, moderately significant differences ($0.001 \leq P \leq 0.010$) for Mg, EK and C:N, and significant differences ($P \leq 0.05$) for K and silt. No significant differences in subsurface soil between vegetation types were found for ECEC, clay or sand. Table 5.2 shows mean values for each subsurface soil variable, and homogenous groups of means among vegetation types.

Table 5.2: Mean subsurface soil variables (20–30 cm) for each vegetation type.

	River red gum (n=13)	Coolibah (n=17)	Grassland (n=17)	Myall (n=6)	Black box (n=4)	Planted (n=4)
ASWAT$^{\#}$	1.00b	9.00ab	9.00ab	9.00ab	6.50ab	12.50a
pH^{1}	7.63b	8.74a	8.73a	8.88a	9.01a	9.30a
EC (dS/m)^{4}	0.07d	0.20bc	0.21c*	0.28ab	0.31ab	0.40a
Ca (cmol+/kg)	22.68a	21.02a	21.42a	18.95ab	19.76ab	13.23b
Mg (cmol+/kg)	12.44ab	14.17a	12.62ab	12.16ab	9.74b	16.78a
Na (cmol+/kg)^{4}	0.88c	2.85ab	2.59b	3.15ab	2.50abc	7.07a*
K (cmol+/kg)^{1}	0.92a	1.13a	0.91a	1.13a	1.48a	0.82a
ECEC (cmol+/kg)	36.91a	39.17a	37.53a	35.38a	33.47a	37.89a
Ca:Mg	1.91a	1.50b	1.75ab	1.60ab	2.06a	0.80c
ECa^{1} (%)	61.75a	53.82bc	57.28ab	53.59bc	59.07ab	34.48c
EMg^{2} (%)	33.38bc	36.09b	33.43bc	34.37bc	29.05c	44.85a
ESP^{2} (%)	2.29c	7.24b	6.93b	8.87ab	7.20ab	18.53a
EK (%)	2.58b*	2.85ab	2.36ab	3.18a	4.69ab*	2.15ab
TC (%)^{4}	1.28a	0.77b	0.69b	0.79b	0.63b	0.91ab
TOC (%)^{1}	1.23a	0.72b	0.66b	0.73b	0.57b	0.79ab
N (%)^{1}	0.12a	0.07b	0.07b	0.08b	0.06b	0.07b
C:N^{2}	10.98ab	11.20ab	10.14b	10.27b	10.94ab	13.59a
Clay (%)^{1}	48.53a	54.92a	52.62a	50.24a	51.29a	56.12a
Silt (%)^{2}	25.31a	17.18b	19.35ab	19.11ab	18.95ab	25.40ab
Sand (%)^{2}	26.16a	27.91a	28.03a	30.65a	29.77a	18.48a

^{1}rank-transformed; ^{2}Ln-transformed (ln(X+1) where applicable); ^{3}log-transformed (Log(X+1) where applicable); ^{4}Sqrt transformed to attain normality prior to AOV; $^{\#}$median values; *single outlier excluded. Letters indicate significant differences (Tukey's) where $P \leq 0.05$.

5.3.2 Characteristics of Vegetation Types

Analysis of vegetation characteristics showed highly significantly differences between vegetation types for litter biomass, and litter, herbaceous and canopy cover ($P < 0.001$).

No significant difference between vegetation types was found for herbaceous biomass ($P = 0.078$). Table 5.3 shows mean values for each vegetation characteristic by vegetation type, and the presence of homogeneous groups of means across vegetation types. Planted sites had significantly higher litter biomass and cover than other vegetation types while grasslands and myall-dominated vegetation generally had lower litter values. River red gum sites had significantly higher herbaceous plant cover while coolibah-dominated sites had significantly less. Grasslands had significantly lower canopy cover than other vegetation types, which had similar mean canopy cover.

Table 5.3: Mean litter, herbaceous vegetation and canopy cover (%), and litter and herbaceous vegetation biomass (kg DM/ha) for each vegetation type.

	River red gum (n=13)	Coolibah (n=17)	Grassland (n=17)	Myall (n=6)	Black box (n=4)	Planted (n=4)
Litter Biomass ^{1}(kg DM/ha)	3284.40ab	2153.70ab	1476.40c	2431.70c	1488.70b	5880.20a
Litter Cover^2 (%)	48.28ab	30.84abc	19.76d	27.69bcd	11.98cd	73.06a
Herbaceous Biomass^3 (kg DM/ha)	1478.00a	1405.40a*	2078.70a	1140.30a	1150.50a	2135.40a
Herbaceous Cover (%)	43.39a	22.35c	37.12ab	19.48b	20.61b	35.17ab
Canopy Cover^1 (%)	22.42a	23.88a	1.18b	17.50a	17.75a	22.50a

^{1}rank transformed; ^{2}Ln transformed; ^{3}log transformed to attain normality prior to AOV; * single outlier excluded. Letters indicate significant differences (Tukey's) where $P \leq 0.05$.

5.3.3 *Water Stable Aggregation*

Pearson correlations showed that aggregate stability (MWD) was significantly positively correlated with C:N ($r = 0.890$, $P = 0.001$) and TOC ($r = 0.768$, $P = 0.016$) in surface soil samples ($n = 9$; Appendix 2). Although the automated BIC procedure in R included the terms TOC and C:N to model MWD, multiple regression analysis showed TOC was not required to explain MWD ($P > 0.05$). Figure 5.2 illustrates the relationships between (a) MWD and TOC and (b) MWD and C:N.

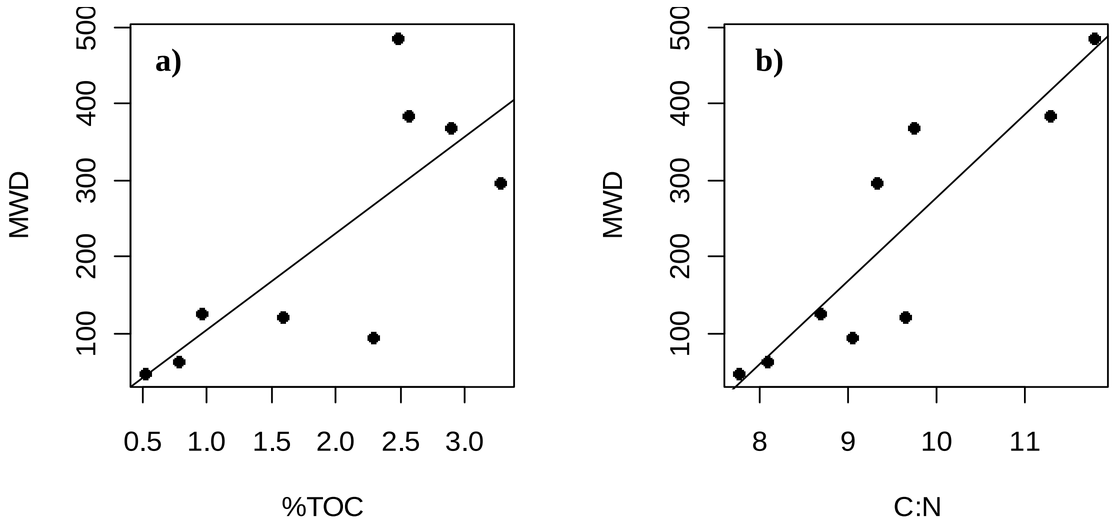


Figure 5.2: Linear regression relationships between MWD in surface soil samples ($n = 9$) and (a) TOC ($MWD = 126.49*TOC - 21.79$ ($R^2 = 0.53$, $P = 0.016$) and (b) C:N ($MWD = 109.08*C:N - 813.41$ ($R^2 = 0.76$, $P = 0.001$).

In subsurface soil samples ($n = 16$), Pearson correlations (Appendix 3) showed no significant correlations between MWD of aggregates and measured soil variables. Stepwise regression using BIC included the terms log (TOC+1), C:N, K, Ca, EMg and log (EC+1) to explain MWD. Multiple regression analysis showed that log (EC+1) was not significant ($P > 0.05$) and not required to explain MWD. The final model used to explain subsoil MWD was ($R^2 = 0.5705$, $P = 0.015$):

$$MWD = 797.02*\log(TOC+1) - 83.69*C:N + 314.24*K - 25.41*Ca + 29.37EMg - 148.65 \quad \text{(Equation 1)}$$

Diagnostic tests showed that Equation 1 satisfied all assumptions of multiple regression, and Pearson's correlation tests showed that none of the explanatory variables were correlated. Linear regression analyses showed that none of the individual soil variables were significantly related to MWD.

5.3.4 Vegetation Influence on Water Stable Aggregation

Data from 60 sites showed that surface soil C:N ratio was positively correlated with litter biomass, litter cover and canopy cover (Table 5.4). Herbaceous plant cover and biomass were negatively correlated with surface soil C:N, but these relationships were not significant. Highly significant positive correlations were also found between surface soil TOC and C:N, litter cover and biomass, and herbaceous plant cover. Herbaceous plant biomass showed a non-significant positive correlation with surface soil TOC, as did canopy cover.

In the subsurface soil, C:N ratio was weakly positively correlated with canopy cover, but not litter cover or biomass (Table 5.4). TOC was strongly correlated with both litter and plant cover and weakly correlated with litter biomass. Subsurface C:N ratio and TOC were again highly correlated. Correlations were found between Ca and EC (negative correlation), EC and plant cover (negative), K and plant cover (negative), and EMg and litter cover (positive). A highly significant negative correlation was observed between TOC and EC in subsurface soil.

Spearman's rank correlations (Table 5.5) showed that litter cover and biomass were positively correlated, as were herbaceous plant biomass and cover. Both litter variables were also positively correlated with canopy cover while both herbaceous plant variables were negatively correlated with canopy cover.

Table 5.4: Spearman correlations between selected vegetation variables and surface and subsurface soil characteristics influencing water stable aggregation (n = 60).

	Litter		Herbaceous			Subsurface soil variables				
	Biomass	Cover	Biomass	Cover	Canopy	C:N	TOC	Ca	EC	EMg
Litter cover	0.872**									
Herbaceous biomass	–0.213	–0.175								
Herbaceous cover	–0.021	0.113	0.615**							
Canopy cover	0.493**	0.462**	–0.273*	–0.219*						
Surface C:N	0.664**	0.602**	–0.155	–0.103	0.764**					
Subsurface C:N	0.215	0.211	–0.242	–0.197	0.317*					
Surface TOC	0.428**	0.532**	0.075	0.528**	0.234	0.335a**				
Subsurface TOC	0.366*	0.450**	–0.040	0.366**	0.201	0.335**				
Subsurface Ca	–0.056	–0.113	0.199	0.205	–0.094	–0.326*	0.065			
Subsurface EC	–0.225	–0.259*	–0.172	–0.507**	0.069	0.231	–0.437**	–0.534**		
Subsurface EMg	0.245	0.330**	0.033	–0.018	0.232	0.209	–0.007	–0.121	0.039	
Subsurface K	0.053	–0.131	–0.035	–0.408**	0.215	–0.066	–0.220	0.157	0.242	–0.065

[a] surface C:N correlation with surface TOC

*$P \leq 0.05$

**$P \leq 0.01$

5.3.5 ASWAT – Site Level

Spearman's correlation tests revealed ASWAT scores for surface soils (0–5 cm) were highly correlated with ESP and Na (positive), and Ca and ECa (negative; Appendix 4). Weaker but significant relationships were also found between ASWAT and silt, Ca:Mg and C:N (negative correlations). Multicolinearity was shown between most variables correlated with ASWAT (Appendix 4) and a multiple regression model could not be constructed.

Principal components analysis of the surface soil data (Figure 5.3) showed that the first principal component (PC1) accounted for 31.2% of the variance in the data while the second (PC2) accounted for an additional 23.6% (total 54.7%). Figure 5.3 shows median ASWAT scores plotted for 60 sites in relation to measured soil variables and PC1 and PC2. The PCA indicated that low ASWAT scores were associated with surface soils with high ECa, silt, N, TOC+1 and Ca, and negatively associated with soils of high pH, ESP and Na. Variables contributing most to PC1 were ECa and Ca:Mg (positive), and EMg and clay (negative). Variables contributing most to PC2 were pH and ESP (positive) and TOC and Ca (negative). Sites with low ASWAT scores feature most prominently in the bottom half of the plot, though low scoring sites are spread across most of the ordination space.

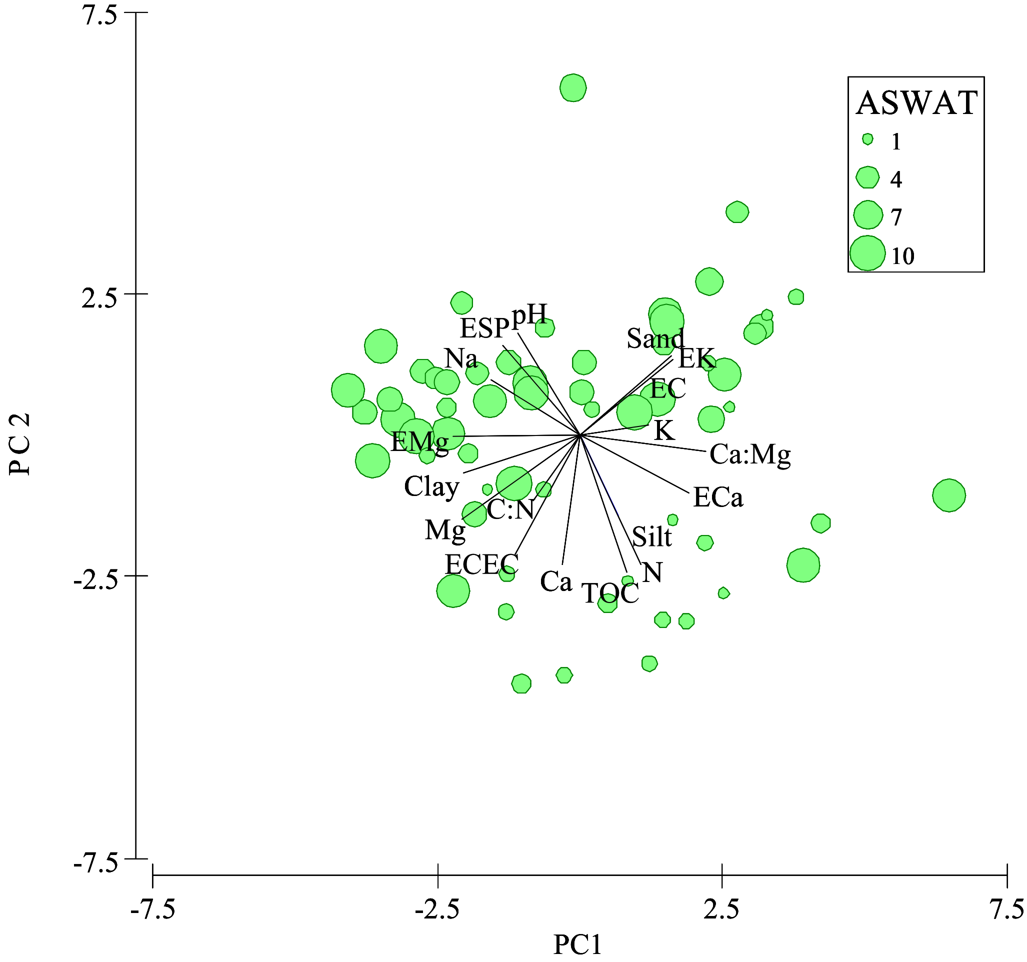


Figure 5.3: PCA of 18 surface soil variables (normalised and standardised) for 60 sites plotted against PC1 and PC2. Bubbles represent the median (n = 9) ASWAT score for each site, larger bubbles indicating more dispersive soils.

Figure 5.4 shows the same PCA with vegetation type plotted as sites and an additional vector representing ASWAT. Sites with low ASWAT scores were predominantly river red gum sites, these sites clustering more tightly than other vegetation types in relation to PC1 and PC2. Tight clustering of vectors representing ESP, pH, Na and ASWAT (Figure 5.4) implied that as these vectors increased, soil dispersion (as measured by ASWAT) also increased. TOC, N and silt opposed ASWAT in the PCA, which implied decreased soil dispersion as these variables increased. Ca and ECa also feature in the lower right quadrant of the PCA indicating their importance to aggregate stability. Soil vectors more or less orthogonal to the ASWAT vector had less influence on soil dispersion.

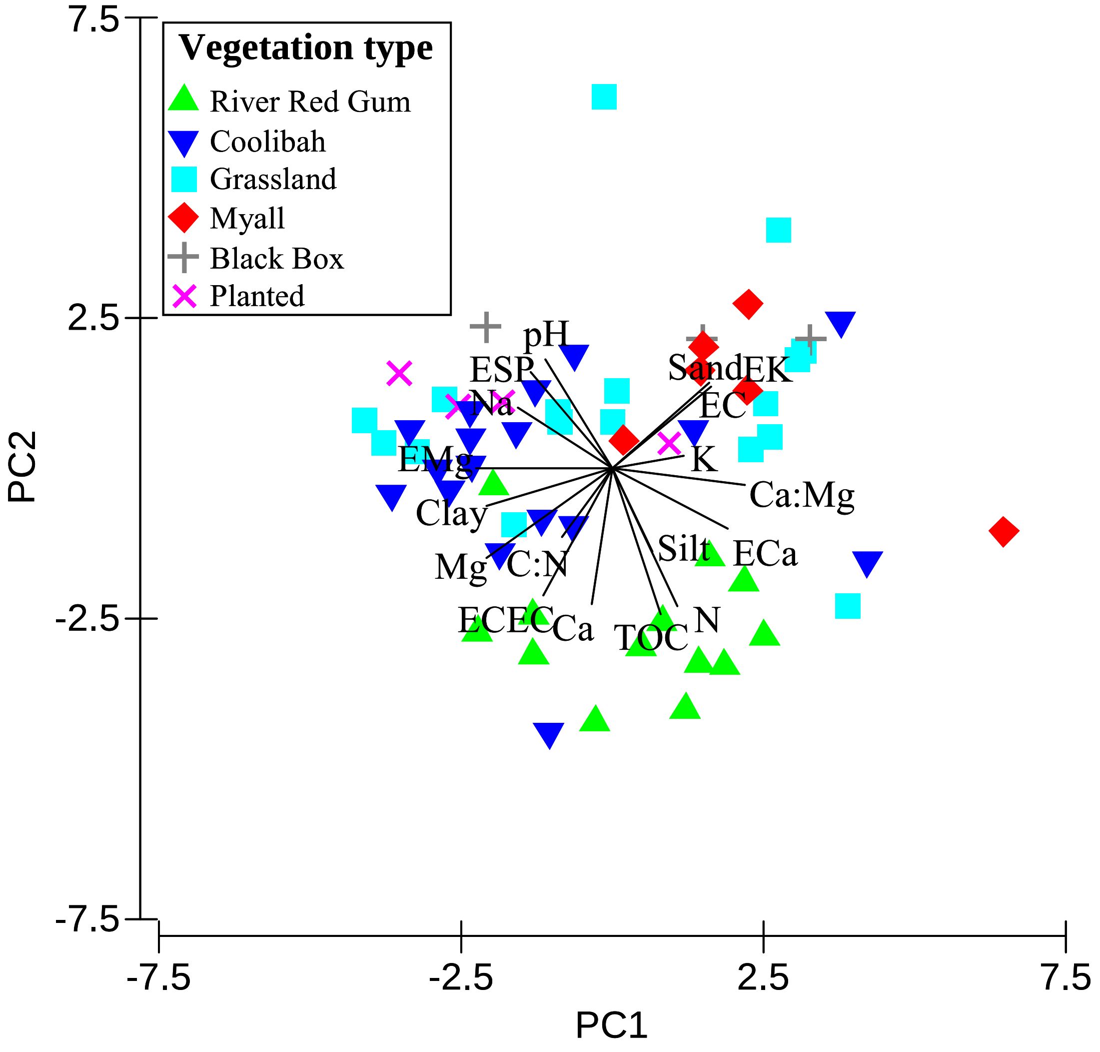


Figure 5.4: Two dimensional PCA ordination of 19 surface soil variables (normalised and standardised) for 60 sites representing six vegetation types commonly found on vertosols on the lower Namoi floodplain.

In subsurface soils (20–30 cm), ASWAT was positively correlated with Na, ESP, pH, EC, clay, Mg and K, and negatively correlated with ECa, N, Ca:Mg, TOC and TC (Appendix 5). Weaker but significant correlations were found between ASWAT and ECEC, and ASWAT and EMg (positive), and ASWAT and silt (negative). Many soil physico-chemical variables that characterised the subsoil showed multicolinearity (Appendix 5). Stepwise multiple regression showed Na was the only variable required to explain subsurface ASWAT scores (Figure 5.5).

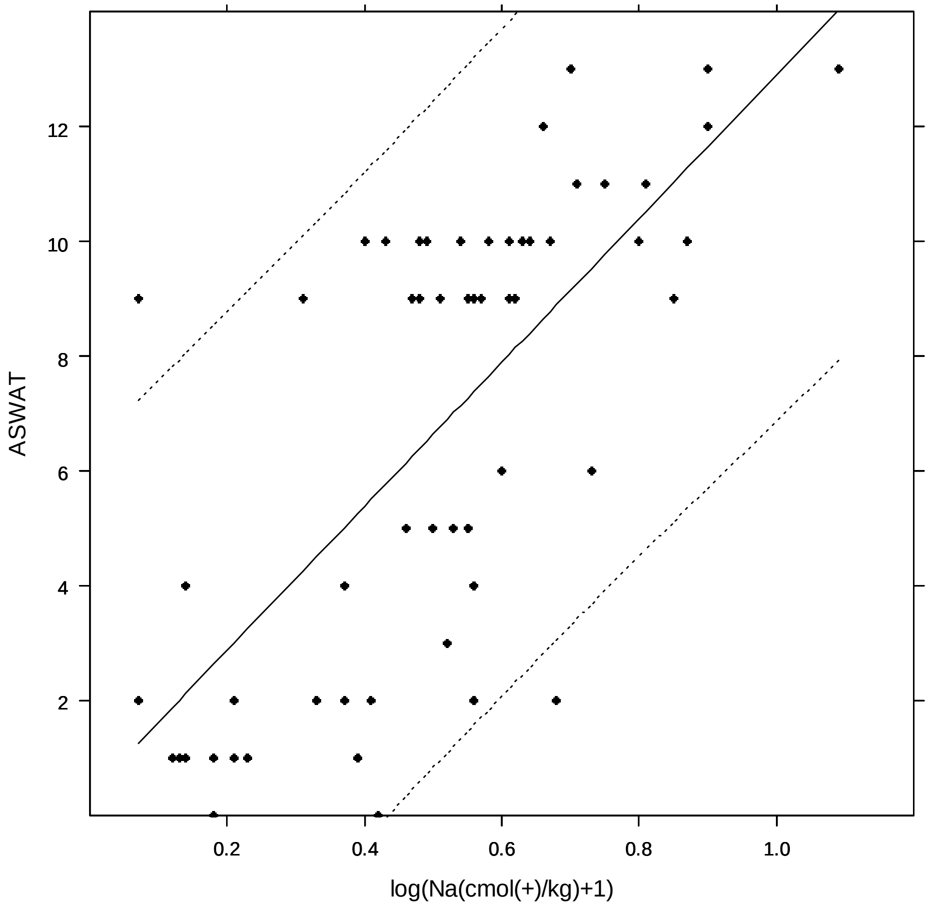


Figure 5.5: Subsurface ASWAT score (n = 60) plotted against log (Na+1) showing the fitted model (ASWAT ~ 12.53*log (Na+1) + 0.375) (R^2 = 0.4953, $P < 0.001$) and 95% confidence intervals.

Principal components analysis (Figure 5.6) was used to reduce the dimensionality of the subsurface soil dataset and highlight variables important in determining soil propensity for dispersion. PC1 accounted for 37.7% of the variance in the data while PC2 accounted for an additional 20.2%. Variables contributing most to PC1 were Na and ESP (positive), and ECa and Ca:Mg (negative). Variables contributing most to PC2 were EC and sand (positive), and ECEC and Ca (negative). Sites with high ECa and Ca:Mg, and low Na and ESP generally featured on the right of the ordination.

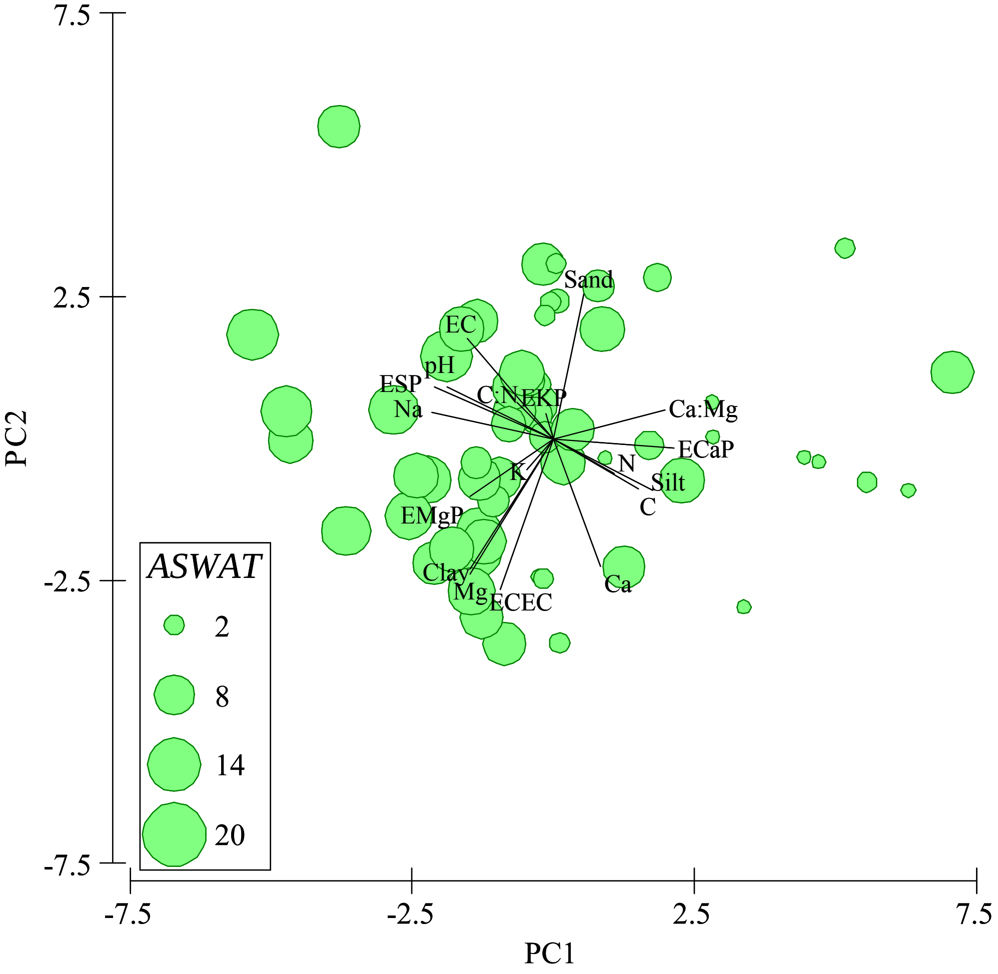


Figure 5.6: PCA ordination of 18 subsurface soil variables (normalised and standardised) and 60 sites against PC1 and PC2. Bubbles represent the median (n = 9) ASWAT score at each site, larger bubbles representing more dispersive soils.

Figure 5.7 shows the same PCA, highlighting the relationship between vegetation type and subsurface soil variables including ASWAT. Figure 5.7 shows that river red gum sites again had the lowest ASWAT scores, while planted and coolibah sites had the highest ASWAT scores. The clustering of subsurface vectors in Figure 5.7 implied that ESP, pH Na and EMg were associated with greater subsurface soil dispersion (as measured by ASWAT). Ca:Mg, ECa, C, N and silt were negatively correlated with ASWAT and were associated with greater subsurface soil aggregate stability.

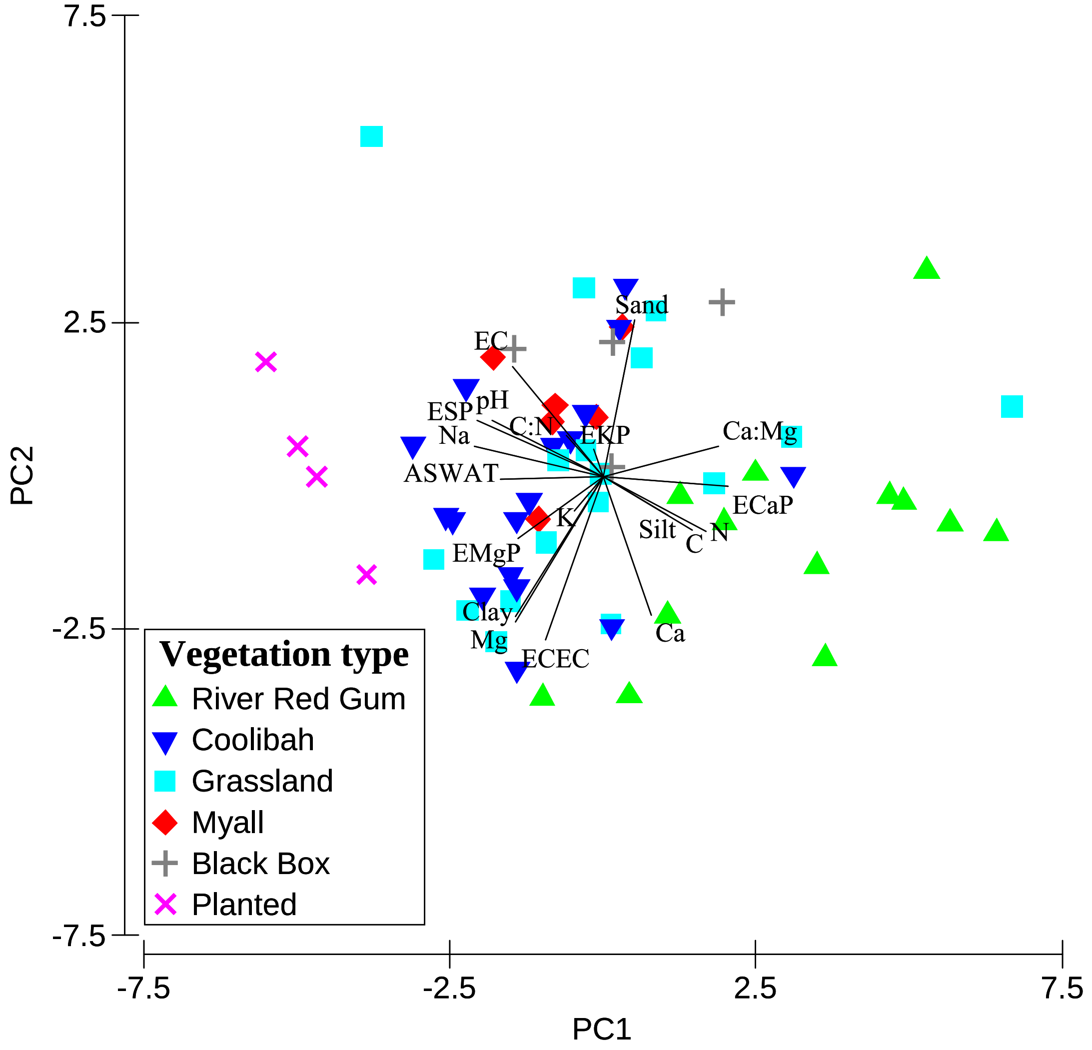


Figure 5.7: Two dimensional PCA ordination of 19 subsurface soil variables (normalised and standardised) for 60 sites representing six vegetation types commonly found on vertosols on the lower Namoi floodplain.

5.3.6 ASWAT – Cover Class Interactions

At the scale of individual soil cores across all vegetation types, the cover classes of chenopods, grasses and graminoids, and bare ground and litter were associated with significantly higher median ASWAT scores in surface soils (Figure 5.8). The shrubs and legumes, and forb cover classes showed significantly lower median ASWAT scores.

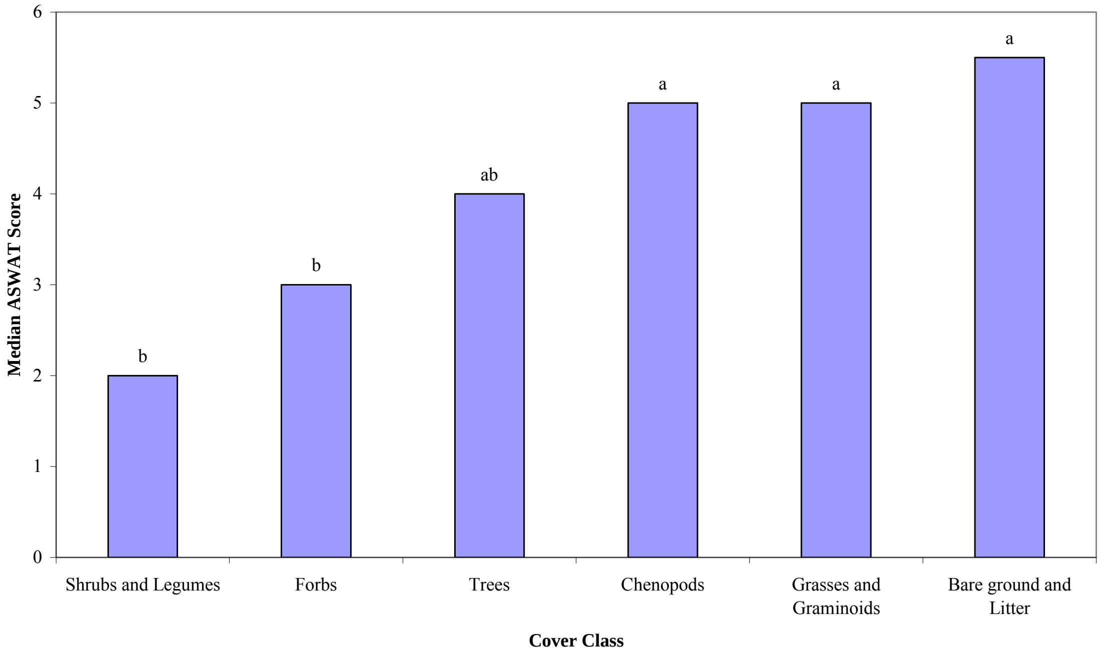


Figure 5.8: Median surface soil ASWAT score for six cover classes ($F_{5,543} = 6.97$, $P < 0.001$). Letters indicate significant differences (Tukey's HSD, $P < 0.05$).

Figure 5.9 shows the proportion of cores within each cover class from least (bottom) to most (top) dispersive across the six vegetation types. When the tree cover class was excluded (not significantly different from either the dispersive or non-dispersive cover class groups; Figure 5.8), the proportion of cores taken from significantly less dispersive (lower ASWAT) cover classes was highest in river red gum sites (55%), and lowest in planted sites (6%) while coolibah, grassland, myall and black box vegetation types showed intermediate values. This result corresponds with Table 5.1, which showed river red gum sites had significantly lower ASWAT scores, planted sites had significantly higher ASWAT scores, and coolibah, grassland, myall and black box sites had intermediate ASWAT scores.

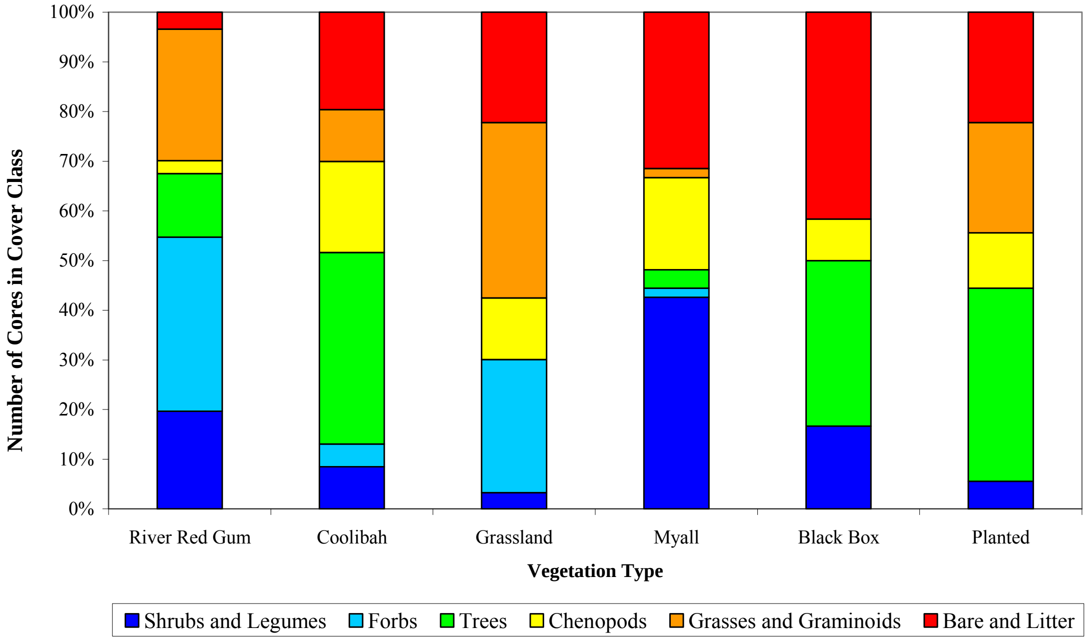


Figure 5.9: Proportion of cores representing each cover class in each vegetation type. Note the unbalanced design of cover classes across vegetation types. Note the unbalanced distribution of both cores and cover classes across vegetation types

5.4 Discussion

Organic matter has been considered less important in determining soil aggregate size distribution and stability in vertosol soils than other soil physico-chemical properties (Charman & Roper 2000; Coughlan 1984; Coughlan et al. 1987; Dalal & Bridge 1996; Prebble 1987; Smith 1984). A degree of aggregate slaking upon rapid wetting of dry aggregates is generally accepted as an inherent trait of vertosol soils (Coughlan 1984; Cass 1999), hence the importance and influence of organic matter has often been overlooked in relation to maintenance of structural stability in these soils. The assumption that vertosol slaking cannot be controlled with organic C is likely to have originated in studies of cultivated land; the majority of cropped vertosols in Australia show a very small range in organic C content, predominantly <2%, an insufficient amount to have an appreciable influence on aggregate stability in these soils (Coughlan 1984).

5.4.1 Water Stable Aggregation

The behaviour and degree of slaking of surface soil upon rapid wetting has major implications in determining soil erodibility, the risk of surface seal development and infiltration rate (Loch 1989). The behaviour of subsurface soils is typically regarded as less important as subsurface aggregates are usually protected from rapid wetting by the surface soil. However, in vertosols that crack to great depths (>1 m) upon drying, particularly just before the summer storm season begins, subsurface aggregates can be exposed to rapid wetting. The stability of subsurface aggregates to rapid wetting therefore has implications for subsurface soil structure, infiltration and erosion. Stability of subsurface aggregates is therefore important in these soils.

The results of this study contradict the widely held belief that vertosol slaking cannot be controlled by organic matter input. The slaking behaviour of the surface (0–5 cm) soil examined in this study was diminished with sufficient organic matter inputs, though the quantity of organic matter alone was less satisfactory in explaining WSA than the quality of the soil organic matter as indicated by the soil C:N ratio. Organic matter in subsurface soil samples was highlighted as an agent controlling slaking, with both TOC and C:N showing significant linear relationships with MWD. However, in contrast to surface soils where both variables were positively related to MWD, TOC was positively related but C:N was negatively related to MWD in subsurface samples.

When organic carbon content reached 2.5–3% in surface soils, 90–100% of aggregates were large enough to be retained in a 250 µm sieve, and macro-aggregate breakdown caused by slaking was reduced. Le Bissonnais and Arrouays (1997) reported that 25–30 g kg^{-1} organic matter (2.5–3% organic matter, approximately 1.4–1.7% TOC) was an important threshold for aggregate stability in a vermic haplumbrept, and that this value was similar to values reported by other authors. The present study showed with a small number of samples that double that amount may be an important threshold in the topsoil of lower Namoi floodplain vertosols in terms of reducing soil slaking. Other authors (see Loveland & Webb 2003) have indicated 2% TOC is an important threshold across a range of soil types, below which general soil quality declines. TOC levels in cropped vertisols are often well below this level, giving rise to the conclusions of earlier authors that vertisol slaking cannot be controlled by organic matter.

Soil C:N measures the degree of decomposition and quality of soil organic matter (Batjes 1996; Coulombe et al. 1996). In the surface soils examined, high C:N was associated with high macro-aggregate stability as reflected in the WSA data. Wilson et al. (2009) reported that arbuscular mychorrizal fungi and glomalin-related soil protein pools associated with mycorrhizal fungi act as organic cements, contributing strongly to aggregate stability. Bossuyt et al. (2001) reported increased soil fungal activity and higher aggregate stability as C:N increased. Changes in microbial community composition could account for the differences in WSA with C:N, though the range in C:N values in this study may have been too small for significant community compositional shift between bacteria and fungi.

Younger, more labile C (termed ‘active carbon’ by some authors) often has high C:N ratios (Jastrow and Miller 1998). This carbon is associated with the rhizosphere of plants and exists in the form of simple sugars or polysaccharides (Lavelle & Spain 2001). This carbon pool when associated with temporary binding agents plays a large role in stabilising soil aggregates (Tisdall & Oades 1982). In bulk soil (i.e. away from plant roots), however, high C:N ratios can be indicative of partly decomposed organic matter after some N has been preferentially mineralised (Spain & Hodgen 1994; Swift et al. 1979). The source and quality of litter inputs has an overriding effect on the soil C:N ratio, as does the successional stage of the vegetation (Cerdà 1998; Snowdon et al. 2005; Swift et al. 1979).

Soil C:N generally decreases slightly with depth, reflecting the greater humification and age of the C stored at depth (Lavelle & Spain 2001). Batjes (1996) showed that C:N does not decrease with depth in vertosols due to the degree of cracking, mixing and churning typical of these soils. Our results showed very little difference in C:N values between 0–5 cm and 20–30 cm. Reduced biological activity and glomalin with depth (Cass 1999) may explain the change in sign of the relationship between MWD and C:N from positive at 0–5 cm to negative at 20–30 cm depth. In addition to TOC and C:N, soil chemical variables included in the best-fit model explaining subsurface soil stability were K, Ca and EMg. In low organic matter environments, these variables can influence soil physical properties (Pernes-Debuyser & Tessier 2004).

5.4.2 Vegetation Types and Water Stable Aggregation

C:N ratios in this study were lowest in grasslands and highly correlated with tree canopy cover, signifying the important influence of trees on aggregate stability. Tree canopy cover was also strongly correlated with litter cover and biomass which showed strong correlations with C:N, therefore decomposed tree litter may be disproportionately important in soil aggregate stabilisation. The litter in woody plant-dominated ecosystems is more lignified and therefore decomposes more slowly than that in grassland and open woodland ecosystems where litter has a much lower C:N ratio (Swift et al. 1979). The canopies of vegetation types sampled in this study were dominated by sclerophyllous trees, predominantly *Eucalyptus* species. Eucalypt forests have a higher soil C:N than other vegetation types (Snowdon et al. 2005), especially grasslands, and this can be attributed to differences in lignin concentrations (Swift et al. 1979).

Soil TOC content was strongly correlated with plant cover, and litter cover and biomass, and in highest concentration in river red gum soils. *Eucalyptus*-dominated sites sampled in this study, especially river red gum and some coolibah sites, contained very large, old trees, some in excess of 3 m DBH. These trees produce large quantities of litter and much larger quantities of coarse woody debris (high C:N) than other vegetation types, and owing to their large size, have been doing so for much longer than trees at other sites. The woody debris in these sites breaks down slowly (Mackensen et al. 2003), and provides a large sustained input of organic material to soil microbial communities. In contrast, grasslands produce litter that rapidly decomposes and inputs are pulsed in response to seasonal conditions such as summer rainfall or senescence of winter annuals in early spring.

Planted sites had topsoils with higher C:N than river red gum and coolibah sites. Despite having low TOC concentrations, two planted sites, in particular, had high C:N ratios. Both sites have a history of soil disturbance through repeated cultivation or removal of surface soil for construction of a levee bank (J. Watson & M. Logan, pers. comm., 2008), therefore the high C:N could be a result of their land use history. Previous disturbance may have resulted in low N levels at the sites due to more rapid denitrification than other sites (Swift et al. 1979). Also, the young trees and shrubs planted at these sites may have taken up all of the available N which had been reduced by previous disturbance, and litterfall from the new trees and shrubs has so far been insufficient to replace N reserves in the soil (Snowdon et al. 2005).

5.4.3 ASWAT

ASWAT measures the propensity for soil dispersion and was strongly influenced by soil chemistry, and measures of calcium and sodium, in particular, in both surface and subsurface soils. Calcium concentrations (Ca and ECa) were negatively correlated with ASWAT and were therefore associated with reduced dispersion. Measures of sodium (Na and ESP) were positively correlated with ASWAT indicating the role of Na in increasing the potential for soil dispersion. High ECEC and TOC also appeared to be associated with sites characterised by low ASWAT scores as illustrated in the PCA ordinations.

Calcium is a divalent cation and an important determinant of soil structure and aggregate stability (e.g. Emerson 1967; Rengasamy et al. 1984; Rengasamy & Olsson 1991). The Ca^{2+} ion tends to replace sodium on the exchange complex of the clay component in soil if there is a source of Ca^{2+}, and due to its smaller size and smaller affinity for water, reduces the potential for swelling between layers within the mineral structure, especially in smectite-dominated clays (Emerson 1983). Swelling within the clay mineral structure may cause the clay fraction of the soil to disperse. Calcium helps flocculate dispersed clay and is more effective than the other three major cations in this role (Curtin et al. 1994).

Soil Ca content is influenced by organic matter and soil ECEC. As both organic matter and clay content increase, so too does soil surface area and hence the ECEC. The positive relationship between Ca and clay, and the negative relationship between Ca and sand found in this study reflects these physico-chemical relationships. Soil organic matter has the highest ECEC of any soil constituent, up to 500 cmol(+)/kg (McKenzie et al. 2004). Hence, it is not surprising that Ca was correlated with TOC and C:N in surface soils. Soils with a higher proportion of smectite clays also have a higher surface area and therefore, ECEC (McKenzie et al. 2004). The soils in this study have high smectite contents compared to illite and kaolinite concentrations (Field 2000). Calcium carbonate ($CaCO_3$) is another source of Ca^{2+} in soils, conferring protection against dispersion (Chorom & Rengasamy 1997). While not measured directly in this study, the reaction with orthophosphoric acid when it was added to samples to remove inorganic carbon from samples prior to measurement of organic carbon indicated $CaCO_3$ presence.

Sodium is the largest of the four common cations (Ca^{2+}, Mg^{2+}, K^+, Na^+) in soil and, being well hydrated and monovalent (compared to the divalent Ca^{2+}), it is held more weakly by negatively charged clay surfaces (Rengasamy & Churchman 1999). Soils with sufficient Na^+ as a proportion of the soil ECEC, or a high exchangeable sodium percentage (ESP ≥ 6%), are said to be sodic and many are highly dispersive on wetting (Northcote & Skene 1972). Australia has the largest proportion of sodic soils of any continent, much of it concentrated in agriculturally productive soils such as the cracking clays of Victoria and NSW (Rengasamy & Churchman 1999). Sodic soils can occur naturally due to weathering of Na-rich parent materials, or as a result of anthropocentric activities such as land clearing or irrigation with saline water. Sodicity causes or enhances the swelling and disintegration of soil macro-aggregates which may lead to clay dispersion, thereby leading to deterioration of soil physical condition (Sumner 1993). At depth, sodium and divalent cations are the primary determinant of soil structural stability. Subsurface sodicity is an inherent property of many dryland soils across Australia (Rengasamy 2002) especially in the cracking clay soils that make up a large portion of the lower Namoi floodplain (Namoi CMA 2009). Sodic subsoils are amongst the most erodible soils when exposed to rapid wetting, with extensive gully and tunnel formation (Doyle & Habraken 1993). This form of erosion is particularly destructive and expensive to repair hence preventative action, especially in agricultural environments such as the lower Namoi floodplain is required to avoid this type of damage.

The reduction in organic carbon content of soils associated with farming cause significant variation in soil ionic composition of soils compared to undisturbed soils (Naidu et al. 1996). Native vegetation dominated by structurally and biochemically different plant life-forms is likely to differentially affect organic matter dynamics in a similar manner to soil properties determined by physico-chemical variables, with consequent effects on soil stability. The results of this study indicate that organic matter may influence soil dispersive behaviour in these vertosols through its influence on ECEC. Organic matter has a direct influence on soil CEC and, consequently, on exchangeable cations, but, this relationship is highly dependent on soil pH: alkaline soils, such as the vertosols in this study, gain 45% of their CEC from organic matter (Helling et al. 1964). Oades (1995) reported that interactions between polyanionic organic materials and variable charge oxides creates a variety of chemical and physical properties, and that the loss of organic materials with cultivation can thus influence cation and anion exchange, adsorption of phosphate, acidity and clay dispersion. In addition to influencing ECEC directly through organic matter inputs as described earlier, trees influence soil chemistry through litterfall, throughfall and stemflow.

5.4.4 Cover Classes

Surface soils beneath forbs, and shrubs and legumes were associated with significantly lower ASWAT scores than other patch cover classes, while grasses and graminoids, chenopods, and bare ground and litter were associated with significantly higher ASWAT scores in the topsoil beneath all native vegetation. Soil chemistry and texture analyses were not undertaken on individual cores, only as bulked samples across each site, so independent evidence of the mechanisms responsible for the differences in

ASWAT score is unavailable at this stage. However, the dominant shrubs and legumes were *Acacia* (e.g. *Acacia stenophylla*, *A. salicina* and *A. pendula*), suggesting that nitrogen fixation or the enhanced biological activity under these shrubs has a significant effect on soil dispersion, probably as a result of increased nutrient cycling and soil organic matter accumulation beneath these plants. Fertile haloes of soil beneath leguminous shrubs in semi-arid and arid environments have been described numerous times throughout the world (e.g. Charley & Cowling 1968; Facelli & Brock 2000). Chenopods accumulate salt in the surface (0–15 cm) layers of soil under plant canopies, affecting soil physico-chemical properties (Sharma & Tongway 1973) and explaining the influence of this cover type on the increased propensity for soil dispersion beneath chenopods.

Bird et al. (2002) reported strong spatial heterogeneity in C, N and C:N in semi-arid rangeland communities: C, N, C:N and glomalin were highest under mesquite (*Prosopis glandulosa*) and lowest in interspaces, while black gramma (*Bouteloua eriopoda*) (a grass) had intermediate values. The heterogeneity in C, N and C:N concentrations was reflected in the aggregate stability of soils beneath the different cover classes in the present study. Naidu et al. (1996) reported similar variability in organic carbon content in *Eucalyptus* dominated sites, with values ranging between 1.5% and 3% TOC depending on where the soil core was placed in relation to different cover classes. They also reported greater aggregate stability under *Eucalyptus*-dominated woodland than grassland, as was the case in this study.

Increased organic carbon content is a potential mechanism contributing to greater aggregate stability under forbs, and shrubs and legume cover classes. This may be due to water repellance of soil aggregates as a result of increased organic carbon content derived from particular parent sources, such as fungal hyphae (Amézketa 1999). By reducing the ability of aggregates to take up water and the rate of water uptake, organic matter could reduce slaking in the soils in this study. The importance of organic matter in reducing clay dispersion is not as well understood as the role of Na and Ca, though it is acknowledged (Emerson 1954; Emerson 1983; Rengasamy et al 1984). Further investigation of the specific mechanisms responsible for differences in dispersion in soil is required.

5.4.5 Influence of Vegetation and Different Vegetation Types on Soil Aggregate Stability

Vegetation was an important determinant of aggregate stability in these vertosols, in particular macro-aggregate stability. In both surface and subsurface soils, C:N and TOC were the most influential in reducing slaking of aggregates. Tree canopy, litter and herbaceous plant cover, and litter biomass were positively correlated with C:N and TOC, illustrating the importance of these vegetation components in increasing soill structural stability and mitigating erosion. The tree component of the vegetation in this study was more important than herbaceous vegetation in providing large amounts of litter with high C:N ratio and hence contributes disproportionately to erosion mitigation.

The soils in river red gum communities showed greatest aggregate stability. Most of these sites had high soil organic carbon contents compared to other vegetation types and the organic matter had a high C:N ratio. Riparian sites are biologically active areas, particularly in semi-arid areas, and can have significantly higher net primary productivity (NPP) compared to ecosystems in the surrounding landscape (Naiman et al. 2005). The riparian sites in this study were dominated by large old trees that had likely been present and influencing soil properties for centuries. In the case of such trees, even long after an individual tree dies, its influence on soil properties continues for some time as the accumulated organic matter slowly breaks down and decomposes (D. Tongway, pers. comm.). In terms of a lasting contribution to erosion mitigation, river red gum communities are invaluable.

5.4.6 Management Implications

Soils under vegetation with a high litter input, especially where this litter had a high C:N ratio, showed greater aggregate stability than soils subject to less litter input. High C:N ratios were found in sites with a high canopy cover of *Eucalyptus*, namely river red gum sites. Litter management and, in particular, incorporation of *Eucalyptus* litter into soil should therefore increase macro-aggregate stability. Litter decomposition also increases soil ECEC and macro-faunal community activity, which in turn increases soil resistance to dispersion. Planted vegetation sites sampled in this study showed significant litter inputs and further research is required to quantify the potentially beneficial effect of plantings on soil aggregate stability, and over what time scale this takes place. Well-managed grassland sites provide significant litter inputs to soil, but as grass litter has a lower C:N ratio (Lavelle & Spain 2001), greater inputs of this litter will be required to achieve the same effect as *Eucalyptus* litter. Woody-dominated communities have higher NPP than grassland communities and presumably a greater accumulation of litter in the soil than grasslands. It is therefore unlikely that grasslands will generate the quantity and quality of litter produced by woody-dominated communities and are hence less able to contribute to soil aggregate stability and erosion mitigation. Persistence of litter inputs year-round may also be important in providing a more constant source of food and habitat for soil micro-organisms. Litter inputs in semi-arid grasslands are likely to be pulsed as a result of sporadic rainfall and drought. Further research is required to determine the influence of individual plant types and bare soil on clay dispersion, as the results of this study indicate that plant types alter soil behaviour to the extent of stopping or slowing clay dispersion.

5.5 Conclusion

Variation in surface and subsurface aggregate stability to slaking occurred in vertosols supporting different native vegetation types and this was predominantly related to TOC and high C:N litter availability. Slaking is not an inherent trait of vertosols. As is the case with most other soil types, the degree of slaking in vertosols as measured by macro-aggregate stability in water can be reduced through input of sufficient high quality (i.e. high C:N ratio) organic matter. With regards to clay dispersion, little evidence was found that vegetation or vegetation management can overcome the effect of sodium in subsurface soils, but organic matter through its influence on soil chemistry altered the susceptibility of soil to dispersion and hence erosion in surface soils.

Chapter 6: Biodiversity Conservation Value: Variation with Vegetation Type and Grazing Management on the Lower Namoi Floodplain, NSW

A white-breasted sea eagle leaving its perch high in a river red gum (*Eucalyptus camaldulensis*) tree on the edge of the Namoi River at Walgett (June 2009).

Chapter 6: Biodiversity Conservation Value: Variation with Vegetation Type and Grazing Management on the Lower Namoi Floodplain, NSW

6.1 Introduction

Land use change the world over has led to rapid alteration of the biosphere, including changes in the composition, structure and function of many ecosystems and impacts on their ability to provide ecosystem services (MA 2005). Clearing of native vegetation on privately owned land in cropping districts across Australia has been extensive in the past 100 years (Benson 1991; Cox et al. 2001; Hobbs & Hopkins 1990; Metcalf et al. 2003). Native vegetation now predominantly occurs as isolated remnants in fragmented landscapes (Hobbs 2001). In many areas, remaining native vegetation is grazed by introduced livestock for economic gain, and this has further contributed to vegetation changes, with implications for biodiversity (e.g. Benson 1991; Hobbs 2000; Jansen & Robertson 2001; Yates et al. 2000). Native vegetation remnants on private land have the potential to contribute to biodiversity conservation and as such, incentives for management and markets for biodiversity credits are now emerging (e.g. environmental stewardship payments, conservation auctions: Windle et al. 2009 and Biobanking: DECC 2007). In order to determine the value of native vegetation on private agricultural land for biodiversity conservation, a detailed assessment of the biodiversity existing on private and public land in agricultural areas and of the impacts of prevailing management is necessary.

6.1.1 The Value of Native Vegetation in Agricultural Landscapes

Landholders in agricultural landscapes, especially areas extensively developed for cropping, often do not know the value of their native vegetation for biodiversity conservation, describing it as just ‘the bush block’ or ‘the horse paddock’. In highly modified environments, such remnants become exceedingly valuable as they constitute the last remaining habitat or refuge for many species (Tscharntke et al. 2005). Past conservation efforts have centred on management of large ecological reserves, but the success of such efforts hinges on surrounding land use and connectivity (Wallington et al. 2005; Pimentel et al. 1992; Tscharntke et al. 2005). In addition to its outright value alone, remnant vegetation in agricultural areas links larger conservation areas and thereby complements the formal conservation reserve estate (Margules & Pressey 2000). Information on the biodiversity conservation value of native vegetation in modified environments is required to provide a platform for development of effective and realistic conservation and ecosystem restoration strategies, and to demonstrate the value of remnants to landholders (Fischer & Lindenmayer 2002). As land use decisions are made at the individual site or paddock scale, biodiversity conservation value also needs to be assessed at this scale to be useful to individual landholders.

6.1.2 Biodiversity in Agricultural Landscapes

With realisation of the importance of biodiversity in providing ecosystem services that sustain agriculture, greater attention is now paid to the amount and distribution of biodiversity in highly modified agricultural landscapes (McNeely & Scherr 2003). Much of the world’s biodiversity exists outside conservation reserves in human managed systems (Burbidge & Wallace 1995; Pimentel et al. 1992). However, agriculture and biodiversity conservation are often considered incompatible land uses, and there is a history of inability to integrate biodiversity into agricultural practice (Salt et al. 2004; Tscharntke et al. 2005). While agriculture relies heavily on biodiversity, its success has increasingly resulted in loss of ecosystem complexity and biodiversity in many areas due to habitat loss, fragmentation, altered disturbance regimes and decreased ecosystem functioning (Foley et al. 2005; Heywood 1995; MA 2005).

Virtually all taxonomic groups of organisms are negatively impacted to varying degrees as a result of landscape modification and habitat fragmentation (Foley et al. 2005; Fischer & Lindenmayer 2007). As landscape modification increases, native vegetation is lost, land use intensity increases, and remnants of native vegetation are increasingly put under pressure by processes originating in surrounding lands (McIntyre & Hobbs 1999). Arable cropping land has been impacted to a major extent with less than 10% of the original vegetation remaining in some areas (Williams et al. 2001).

A great deal of effort has been dedicated to developing theories to explain habitat loss and fragmentation effects at a regional scale (e.g. pattern-oriented approaches: Fischer & Lindenmayer 2006) and designing reserve systems based on ideal patch size, shape, configuration (Saunders et al. 1991). While theories provide a basis for further research, they have proved of little practical value to land managers responsible for managing individual remnants in fragmented landscapes (Saunders et al. 1991). Few data are available to determine which habitats are important, classify priority sites for management action, and identify species likely to be lost from individual remnants in modified landscapes. At the other extreme, study of the impacts on individual species of processes arising from landscape modification ('species-oriented approach:' Fischer & Lindenmayer 2006) are justified on the basis that species react individualistically to habitat fragmentation. However, it is not feasible to study every species to the level of detail required to be useful for management. Accordingly, general principles have been developed from studies of the impact of landscape modification on different taxonomic groups (Fischer & Lindenmayer 2007; Hobbs & Yates 2003).

The response of plant species to landscape modification and habitat fragmentation was reviewed by Hobbs and Yates (2003). They concluded that the dominant factors influencing plant species responses to landscape modification and fragmentation were life-form (a small remnant for a tree may be a large fragment for a herb), natural abundance (rare species are more susceptible to local extinction than common species), reproductive dependence on species mutualisms (reliance on one or few pollinators or dispersers makes a species more susceptible to extinction than interactions with many pollinators or seed dispersers), breeding systems (self-incompatible species are more vulnerable to extinction than self-compatible species), reproductive dependence on seeds (vegetatively reproducing species are less vulnerable to extinction as they are not affected by factors influencing seed production and seedling recruitment), and resilience and dependence on disturbance for recruitment and competitive ability.

The time-scale over which a species is affected by habitat fragmentation is dependent on the life-cycle or life-form of the species. Longer-lived species can persist in a landscape long after land use change, but may be unable to recruit and hence will not survive the current generation (Hobbs & Yates 2003). Short-lived species can reproduce and complete their life-cycle quickly under favourable conditions. Hence, in the case of plants, if seed remains viable and favourable conditions eventuate, these species are more likely to persist in modified environments (Henle et al. 2004). Longer-lived species with persistent seed reserves or the ability to propagate vegetatively may be less susceptible to extinction than similar species relying purely on seed for replacement (Henle et al. 2004). Plant species requiring the services of other organisms, such as birds, to facilitate dispersal are likely to be the most affected by landscape modification, whereas species that are wind or water dispersed are likely to be less affected.

Birds are often used as indicators of vertebrate biodiversity due to the ease with which they are censused and their sensitivity to changes in environmental conditions (Freudenberger & Drew 2001; Jansen & Robertson 2001; Walker & Reuter 1996). Particular groups of birds are more susceptible to landscape modification than others and, as with plants, this may be due to factors relating to the biology of individual species, or external influences associated with the type and extent of landscape modification. In a review of threatened and declining birds in the wheat–sheep belt of south-eastern Australia, Reid (1999) found that most threatened and declining birds were characterised by being ground or low-shrub feeders and dwellers. Only three species listed as declining were canopy foragers and two were aerial insectivores. Both insectivores and granivores were well represented amongst the declining woodland birds, but fewer insectivores were considered threatened. Threatened species typically had naturally low abundances or a restricted range, had a relictual distribution and were often top predators. Declining species once held large ranges across Australia, but as a result of landscape modification, are restricted to smaller areas which cannot satisfy basic ecological or behavioural requirements. Reid (1999) noted that selective clearing of certain vegetation types on fertile soils was the ultimate cause of declining bird numbers, and the ongoing decline of habitat quality of the remaining vegetation would continue to threaten populations. In addition, birds with poor dispersal capabilities were likely to suffer as a result of the interaction between drought or other stochastic events and fragmentation in highly modified environments.

6.1.3 Grazing Impacts on Plant and Bird Communities

Livestock grazing is the most extensive land use across Australia (NLWRA 2002) and the world, especially when associated indirect land uses such as cropping for fodder production and other feed sources are considered (FAO 2002). Changes in ecosystems following grazing are widespread and profound, only rivalled in extent and impact by mining, cropping and clearing (Benson 1991; Wilson 1990). Herbaceous biomass removal is the primary short-term impact of grazing, with flow-on effects for both the plant community and other animals. The long-term effects of grazing depend on grazing intensity and selectivity, sensitivity of plants to defoliation, and interactions with soil, climate and other disturbances (Ludwig et al. 1997; Wilson 1990; Yates et al. 2000).

Grazing of domestic livestock is widely cited as a degrading force in native vegetation remnants (Hobbs 2000; Lunt 1991; Moore 1970; Prober & Thiele 1995; Yates et al. 2000). Impacts can be a direct result of the livestock, or indirect as a result of associated management practices (for example through fertiliser amendment, fire exclusion or the introduction of nitrogen-fixing legumes). The degree of change as a result of grazing depends on the ecosystem. For example, semi-arid woodlands are less resilient to grazing-induced change (Harrington et al. 1984) than Mitchell grass grasslands (Orr 1980). Documented impacts on community composition include increased exotic species richness, cover and abundance (Hobbs 2000; Yates et al. 2000), changes in species richness (McIntyre & Lavorel 1994; Olff & Ritchie 1998; Prober & Thiele 1995; Tremont & McIntyre 1994), decreased rare species richness (Landsberg et al. 2003) and species diversity (Pettit et al. 1995), and extinction of palatable species (Barker & Lange 1969; Hacker 1987).

In addition to changes in community composition, vegetation structure is also impacted by livestock grazing. Grazing impacts vegetation structure through removal of herbaceous biomass (Fondell & Ball 2004), and increases (Harrington et al. 1979; Pressland 1984) or decreases (Dorrough & Moxham 2005; Reid & Landsberg 2000) in woody vegetation cover. Grazing also alters ecosystem functioning in some vegetation communities through impacts on recruitment (negatively by removal of reproductive structures and positively by dispersal of seed and introduction of propagules: Hobbs 2000), fire and other disturbance regimes (Wilson 1990; Yates et al. 2000), erosion (Fanning 1994; Ludwig et al. 1997; Yates et al. 2000), competition (Olff & Ritchie 1998), many soil physico-chemical variables (Greenwood & McKenzie 2001; Yates et al. 2000) and nutrient cycling (Greenwood & McKenzie 2001; Milchunas et al. 1988; Yates et al. 2000). Changes in ecosystem functioning have implications for vegetation persistence and ecosystem service provision.

Grazing of remnant woodlands has been implicated in the decline of many bird species throughout the wheat–sheep belt of south-eastern Australia (Reid 1999). As habitat structure is a major determinant of bird community composition (Jansen & Robertson 2001; MacArthur & MacArthur 1961), one of the greatest impacts on birds as a result of livestock grazing is alteration of habitat, especially modification of understorey vegetation (Dambach 1944; Dambach & Good 1940; Good & Dambach 1943). Grazing can increase or decrease vegetation complexity, thus impacting the availability of nest sites (e.g. Bowen & Kruse 1993; Fondell & Ball 2004) and foraging opportunities (Martin & Possingham 2005). Where vegetation complexity is reduced as a result of grazing, large or aggressive birds may increase in abundance and, through intimidation, competition or predation, drive out smaller bird species (Bently & Catterall 1997; Dow 1979; Grey et al. 1997; Loyn 1987; Piper & Catterall 2003).

6.1.4 Conservation Assessment

Prioritisation is necessary for efficient and effective use of resources for conservation (Margules 1989; Murdoch et al. 2007; Noss & Harris 1986; Pressey et al. 1993; Wilson et al. 2007). Policies that place equal value on all species and regions are ecologically unsound and tactically unachievable given limited resources and competing land uses (Burgman et al. 2001; Possingham & Wilson 2005; Walker 1992). Prioritisation of sites for conservation is akin to the concept of triage in health care and ecological restoration (Hobbs & Kristjanson 2003). Approaches to and priorities for conservation vary depending on scale (i.e. global, continental, country, landscape and site).

The overarching goals of conservation planning are representation (all relevant features are adequately accounted for) and persistence (the processes that ensure the persistence of relevant features are accommodated) (Margules & Pressey 2000). Prioritisation of resources in order to achieve the conservation goals is based on complementarity, irreplaceability and vulnerability. Complementarity hinges on the notion that if the goal of conservation is to represent as much biodiversity as possible over a given area, site differences based on biotic content need to be maximised (i.e. the most dissimilar sites are conserved; Kirkpatrick 1983). Irreplaceability refers to the extent to which the loss of a site compromises regional conservation targets (Margules & Pressey 2000). Irreplacebility is often measured in terms of endemicity, sites with greater endemism having a higher priority for conservation than others (Brooks et al. 2006). Vulnerability is defined as the risk of an area being transformed for extractive uses (Margules & Pressey 2000). Wilson et al. (2005) advocated a three-dimensional approach to assessment of vulnerability based on exposure, impact and intensity of threatening processes, though vulnerability is often difficult to measure.

The degree of emphasis placed on complementarity, irreplaceability and vulnerability vary depending on the scale of assessment. For instance, at a global, continental or nation-wide scale, emphasis is placed on complementarity (e.g. achieving a comprehensive, adequate and representative national reserve system: Commonwealth of Australia 1997) and irreplacebility (e.g. identification of biodiversity hotspots: Mittermeier et al. 2004). At the landscape scale, more emphasis is placed on vulnerability and persistence of biodiversity. Conservation planning at the landscape scale is heavily influenced by notions of habitat connectivity, the degree of surrounding landscape modification and the hostility of the matrix or dispersal potential between different patches in the landscape: McIntyre & Hobbs 1999, 2000). At the site scale, emphasis is generally placed on vulnerability (i.e. habitat management), although endangered species are usually a focus for conservation and special interest groups (such as bird watchers) often consider species identity and irreplaceability important.

Several site-based assessments of biodiversity conservation value are used in Australia (for example BioMetric in NSW: Gibbons et al. 2008; Habitat Hectares in Victoria: DSE 2004; BioCondition in Queensland: Eyre et al. 2008; and Bushland Condition Monitoring in South Australia: Croft et al. 2009). These assessments do not measure biodiversity directly, instead they measure the habitat value of the site based on the assumption that high habitat value (often in comparison with pre-European benchmarks of ecosystem attributes) equates to high biodiversity value (akin to the 'Field of Dreams' notion, 'if you build it they will come:' Hilderbrand et al. 2005; Palmer et al. 1997). These approaches emphasise landscape variables (connectivity, context, patch size etc.) as part of the assessment, but most do not produce information on composition (in terms of species), although they may include the number of species in various life-forms without naming them. While the variables describing landscape context in such assessments provide essential information for catchment management authorities, they are of little value to the individual landholder as farmers generally have little influence over the management of neighbouring lands, and funds are not yet available to undertake extensive farmscape restoration for biodiversity conservation on private land.

6.1.5 Indicators of Biodiversity Conservation Value

The context-specific nature of biodiversity conservation value assessments means that there are numerous indicators in use. Direct measures of biodiversity conservation value are based on attribute richness (number) and diversity (abundance). Depending on the context, information needs and uses, commonly measured attributes include ecosystems, functional groups, genera, all species, rare species, endemic species, exotic or native species and genes. Traditional diversity measures include the Shannon–Wiener and Simpson's indices, but multivariate methods based on data dispersion in multivariate space are commonly used today (Anderson et al. 2008).

Pressey and Nicholls (1989) tested the efficiency (contribution to the conservation of all natural environments or species) of criteria reflecting different measures of diversity and rarity of attributes. Eight individual criteria, four combinations of criteria and three iterative approaches for prioritising features for conservation ('wildlife conservation evaluation': Usher 1986) were investigated. The iterative procedures produced the most efficient priority ranking of sites, but efficiency varied greatly between criteria. Margules (1981, cited in Usher 1986) reviewed the literature on the use of different criteria for conservation evaluation and found that diversity (of species or habitats etc.) was the most widely used criterion, followed rarity, area, threat of human interference. Other criteria included amenity and education value, representativeness, scientific value, recorded history, population size, typicalness, ecological fragility, position in ecological or geographical unit, potential value, average rarity, archaeological interest, availability, importance for migratory wildfowl, management factors, irreplaceability, silvicultural gene bank, successional stage and wildlife reservoir potential (Usher 1986).

6.1.6 Aims and Objectives

Biodiversity is required for the continued supply of ecosystem services, but in many agriculturally dominated regions across Australia, little detailed information exists on the remaining biodiversity, and even less about what was there before, the value of different vegetation types and management histories, and priorities for biodiversity conservation. Much work has been done to determine indicators of biodiversity conservation value at the site level, but little opportunity to apply site assessment tools or validate them in most regions has contributed to a lack of knowledge of the biota remaining in the landscape and their value for biodiversity conservation, especially on private land.

The aim of this study was to assess the biodiversity conservation value of different vegetation types in diverse condition states on the lower Namoi floodplain in northern NSW, Australia. Five vegetation types were included in the study: (1) *Eucalyptus camaldulensis*-dominated woodlands; (2) *Eucalyptus coolabah* woodlands; (3) *Eucalyptus largiflorens*-dominated woodlands; (4) *Acacia pendula*-dominated tall shrublands; and (5) native and derived grasslands. Four objectives relating to the biodiversity conservation value of these vegetation types under different management were pursued in this study:

1. To document the vascular plant and avian species diversity of vegetation communities common on the lower Namoi floodplain;
2. To assign biodiversity conservation value rankings to sites;
3. To identify sites and vegetation types of particular conservation significance based on habitat value and the plants and birds present; and
4. To determine the impact of grazing management on site value for vascular plant and bird conservation.

6.2 Methods

6.2.1 Study Region

The Namoi catchment in northern NSW, Australia, covers 42 053 km^2 and spans four bioregions, the Darling Riverine Plains (DRP), Brigalow Belt South (BBS), Nandewar and New England Tablelands. The Namoi floodplain (Figure 6.1), as defined by the highest recorded flood level, covers an area of approximately 12 000 km^2. The area is characterised by fertile black, grey and brown cracking clay soils that have high agricultural value, especially for irrigated agriculture.

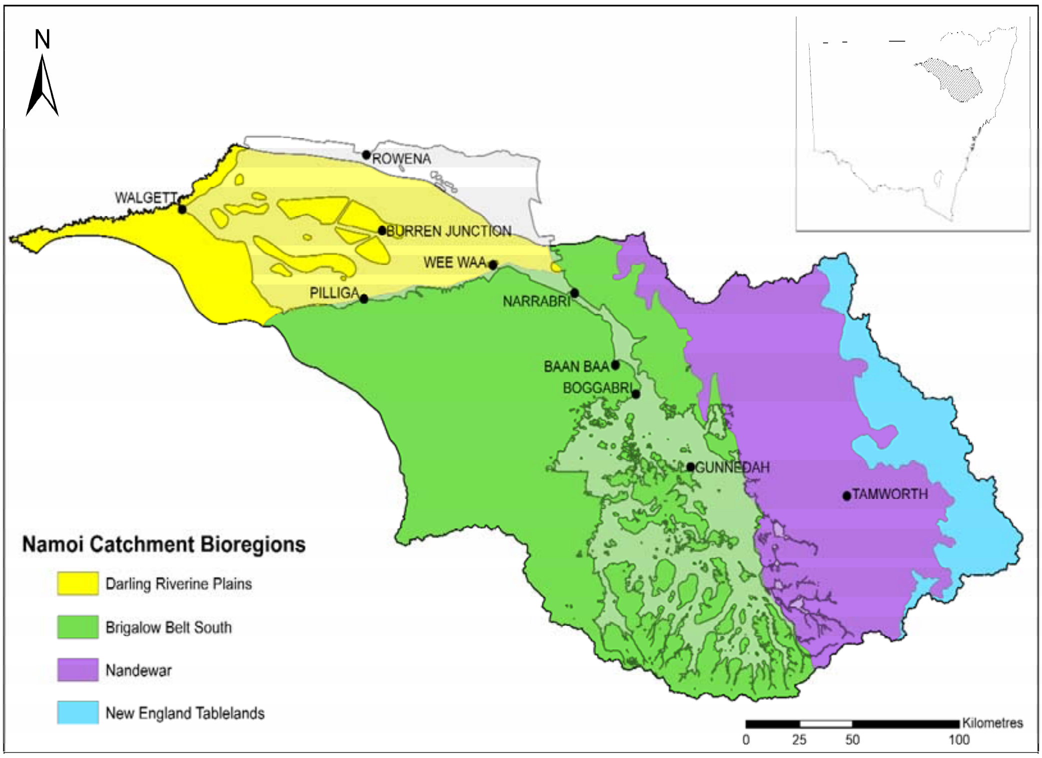


Figure 6.1: Namoi catchment in northern NSW showing IBRA boundaries and the Namoi floodplain.

The floodplain environment has been modified by extensive clearing, changes to hydrological regimes and livestock grazing. Many of the remaining vegetation communities on the floodplain are listed under state (*Threatened Species Conservation*

[TSC] *Act 1995*) or federal (*Environmental Protection and Biodiversity Conservation* [EPBC] *Act 1999*) legislation as endangered ecological communities (EECs). Land use is dominated by agriculture (Figure 6.2). Sheep and cattle grazing, predominantly of native pastures, accounts for 64% of the land use in the catchment, and cropping an additional 20% (Figure 6.2). National parks cover 8% of the catchment, but the distribution of conservation areas is heavily skewed towards steep hill slopes, rocky outcrops and sandy, nutrient-poor soils of little value for agricultural production. The Pilliga National Park, Pilliga Nature Reserve and part of Mount Kaputar National Park are the largest areas set aside for conservation in the Namoi catchment. Very little land has been formally set aside for conservation on the floodplain, though some landholders informally manage small areas for nature conservation (P. Miguel, P. Swansbra and R. Watson, pers. comm., July 2008).

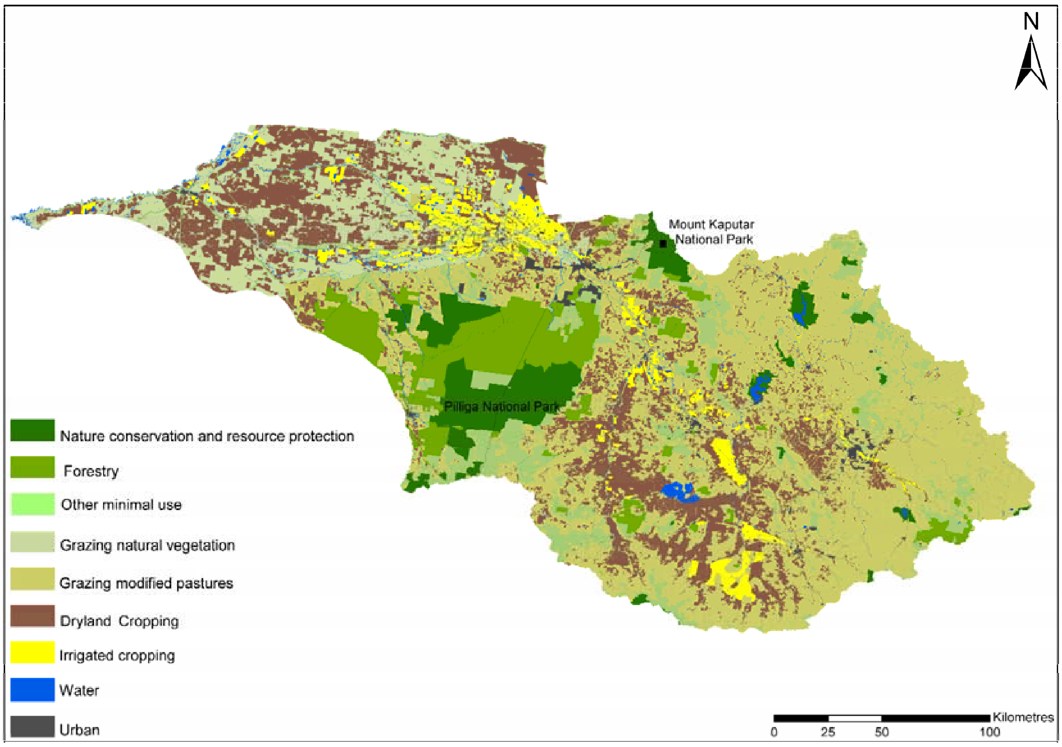


Figure 6.2: Namoi catchment land use showing the extent of agricultural development and location of conservation reserves.

This study was undertaken on cotton farms and travelling stock routes (TSRs) between Boggabri (–30.7000, 150.0333) in the centre and Walgett (–30.0167, 148.1167) in the west of the Namoi floodplain. Rainfall declines to the west, and averages approximately 600 mm p.a. at Boggabri and 400 mm at Walgett. Rainfall is sporadic and influenced by El Niño and La Niña patterns, with slight summer dominance on average. The native vegetation is largely semi-arid to subhumid woodland and grassland, though dense river red gum forests occur along the Namoi River, and densely regenerating coolibah exists in low lying areas on the floodplain. The distribution and structure of the floodplain vegetation is determined by climatic and human driven processes.

6.2.2 Vegetation

The lower Namoi floodplain has a diverse native flora ranging from grassland on large areas of the plains and small areas of chenopod shrubland, through to dense, structurally complex riparian forests along the main river channel. The distribution and persistence of different vegetation types is related to water availability and soil characteristics (Chapter 2), both of which are determined by flood frequency and duration. This study covered the spectrum of environmental and human-induced compositional and structural diversity within and between the major native vegetation types occurring across the floodplain.

6.2.2.1 River Red Gum

River red gum (*Eucalyptus camaldulensis*)-dominated open-forest to open-woodland (Specht 1970) occurs largely as a narrow band of vegetation along the banks of the Namoi River. However, east of Wee Waa, river red gum communities also occur in slight depressions away from the main river channel. Trees away from the main river channel are often smaller in size and experience severe dieback. The cause of the dieback is not known, though several ecological processes are likely to be operating collectively to influence tree health (Reid et al. 2007). Two understorey communities occur in this vegetation type: a grassy understorey with little or no shrub layer dominates sites east of Wee Waa, and a dense, *Acacia*-dominated understorey exists west of Wee Waa.

6.2.2.2 Coolibah

Coolibah (*Eucalyptus coolabah* subsp. *coolabah*)-dominated woodland and open-woodland occurs as a riparian community on minor water courses, sometimes in association with river red gum, and as an open, grassy floodplain community west of Narrabri. Small stands of dense coolibah regeneration (up to 1800 stems ha^{-1} also exist on the floodplain. The ecological community known as *Coolibah-Black Box woodland of the northern riverine plains in the Darling Riverine Plains and Brigalow Belt South bioregions* is listed in NSW as an EEC under the *TSC Act 1995* and was nominated in 2005 as a threatened ecological community (*Coolibah/Black Box Woodlands of the Northern NSW Wheatbelt and Queensland Brigalow Belt*) under the *EPBC Act 1999*.

6.2.2.3 Myall

Myall (*Acacia pendula*)-dominated tall shrubland and tall open-shrubland occurs in slightly elevated areas as small, often isolated individuals and remnants (NSW Scientific Committee 2005). Myall foliage is palatable to stock, especially during drought, and so many trees lack branches <1 m from the ground. In addition, many trees are felled as emergency fodder, further contributing to the decrease in the extent of this community (Beadle 1948). The understorey is dominated by chenopods, although many grasses coexist with the chenopods. *Myall Woodland in the Darling Riverine Plains, Brigalow Belt South, Cobar Peneplain, Murray-Darling Depression, Riverina and NSW South western Slopes bioregions* is listed in NSW as an EEC under the *TSC Act 1995*, and *Weeping Myall Woodlands* are listed as endangered under the *EPBC Act 1999*.

6.2.2.4 Black Box

Black box (*Eucalyptus largiflorens*) woodland occurs on the lower Namoi floodplain west of Burren Junction, though isolated black box trees occur in Coolibah-dominated woodlands across the plains west of Wee Waa. The understorey of this community can be dominated by either grasses or chenopods and a shrub layer of *Eremophila* or *Acacia* spp. may be present. The lower Namoi floodplain is approximately the northern limit of the species as it prefers winter-dominant rainfall; black box replaces coolibah in similar landscape positions further to the south (Beadle 1981). Tree form varies, but small, multiple-stemmed trees are common where the original stem died or was ring-barked and the tree coppiced.

6.2.2.5 Native and Derived Grasslands

Grassland communities on the Namoi floodplain are protected under state and federal legislation: *Natural grasslands on basalt and fine-textured alluvial plains of northern New South Wales and southern Queensland* are listed as a critically endangered EEC under the *EPBC Act 1999* and *Native vegetation on cracking clay soils of the Liverpool Plains* is listed as an EEC under the *TSC Act 1995*. Large areas of grasslands on the floodplain have been replaced by crops, although this land use change occurred on a large scale only since the 1950s with the introduction of suitably powerful machinery capable of cultivating the heavy clay soils (Lang 2008). No area of floodplain grassland is protected in the conservation estate on the Namoi floodplain. The Kirramingly Nature Reserve on the Moree Plains north of the Namoi catchment is the closest conserved area of grassland. Grasslands derived from woodlands as a result of wide-spread ring-barking and thinning activities in the past 150 years are common across the floodplain.

6.2.3 Field Sampling

6.2.3.1 Environmental Characteristics

Soil chemical analyses were used to characterise soils in 54 sites among five different vegetation types. A bulked sample for the 0–5 and 20–30 cm depth increments at each site was passed through a 2 mm sieve using a mechanical soil mill after large pieces of organic matter such as tree roots and leaves were removed. Samples were mixed thoroughly and subsampled for further analyses. Electrical conductivity (EC) and pH were determined in a 1:5 soil-to-water extract. For soils with $pH < 7.5$, exchangeable cations (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}) were determined using atomic absorption spectroscopy following extraction using 1M NH_4Cl. For soils with $pH \geq 7.5$, exchangeable cations were determined at the DPI Victoria laboratory at Werribee using the alcoholic 1M NH_4Cl method (Tucker 1954) after pre-treatment with aqueous ethanol and aqueous glycerol to remove soluble salts. Exchangeable cation values were summed to give the effective cation exchange capacity (ECEC) and Ca:Mg ratio was calculated by dividing Ca by Mg. Exchangeable calcium (ECa), exchangeable magnesium (EMg), exchangeable potassium (EK) and exchangeable sodium (ESP) were calculated as a percentage of ECEC. The soil particle size distribution of samples was determined using the pipette method after overnight end-on-end shaking in sodium hexametaphosphate $(NaPO_3)_6$ dispersant solution (Day 1965) and expressed as proportions of sand, silt and clay using the international classification. A subsample of each bulked soil sample was pushed by hand through a 0.5 mm sieve to remove macro-organic matter and large sand or gravel particles prior to analysis of nitrogen (N) content using a Carlo Erba NA 1500 Solid Sample Analyser.

6.2.3.2 Vegetation

Floristic surveys were carried out in April–May 2008 and October–November 2008. Foliage projective cover of all herbaceous vascular plant species was estimated in permanent 20 × 20 m quadrats, and species were collected for identification in the lab. Foliage projective cover of woody vegetation was estimated in 1 ha plots that encompassed the herbaceous vegetation quadrat. Conditions prior to sampling periods were conducive to maximum plant growth with good summer rainfall (November and December 2007 rainfall was twice the long term average at Wee Waa) encouraging perennial grass growth prior to April–May 2008 and follow-up winter rainfall encouraging germination of winter annuals.

6.2.3.3 Birds

Birds were censused in a 2 ha area of the target vegetation type for 20 minutes during one morning (sunrise to 10:00) and one afternoon sampling period (16:00 to sunset) on different days in September 2007. Weather conditions during sampling were generally fine and sunny (overcast on several mornings) and still or breezy. All birds seen or heard in the 2 ha plot during the census period were recorded along with bird activity (e.g. flying through the plot, perched in a tree, etc.). The dimensions and orientation of plots depended on the configuration of the area to be surveyed but were generally 200 × 100 m. Sites were well separated to minimise overlap of individuals recorded at nearby sites. Birds seen or heard outside the sample plot were noted separately, but those recorded outside the survey area, flying over, or having no obvious interaction with the site (e.g. not hunting, feeding, nesting etc.), were excluded from further analysis so as to standardise survey area and focus on individuals and species reliant on the vegetation. Censuses were carried out by two observers, who each sampled every site once to minimise observer bias.

6.2.3.4 Habitat Characteristics

A modified BOTANAL (Tothill et al. 1978) system was used to estimate vegetation and litter biomass at sites during floristic sampling periods in April–May and October–November 2008. Twenty 50 × 50 cm quadrats located at 4 m intervals around the perimeter of the 20 × 20 m quadrat used for herbaceous floristics were sampled. Biomass of woody vegetation was measured in July 2009 using allometric equations relating DBH or height to total tree biomass, and estimates of coarse woody debris (CWD) biomass were made from diameter and length measurements of fallen timber.

Foliage projective cover was estimated for canopy, tall shrubs (mostly *Acacia* spp. and *Eremophila bignoniiflora*), low shrubs (predominantly chenopods) and herbaceous plants (including grasses, low forbs and graminoids), as were litter and total ground cover during floristic sampling periods. Tree and shrub species richness was also recorded. The number of large trees (>50 cm DBH), small trees (10–50 cm DBH), saplings (<10 cm DBH but taller than 2 m), shrubs (<2 m height), CWD and dead standing trees were also used in habitat assessment at each site.

6.2.3.5 Grazing Intensity

Grazing intensity classes were based on when the site was last grazed and how it had been grazed in the past 10 years. Sites experiencing low grazing pressure had not been grazed in the last 10 years, except by macropods. These sites may have been grazed prior to 2000, but did not sustain large numbers of stock for long periods of time. Sites experiencing intermittent grazing pressure were opportunistically crash grazed within the last 5 years by large numbers of sheep or cattle. Many of the travelling stock routes (TSR sites) were included here. Regularly grazed sites were more often grazed than not by sheep or cattle in extensive paddocks and some well-used TSRs.

6.2.4 Data Analysis

The same analyses were repeated for plants and birds. Five criteria were examined to determine the most efficient way to rank sites in terms of biodiversity conservation value. Native species richness ranks, maximum rarity ranks, and average rarity rankings were calculated for each site, as well as two combinations of the three, ‘species richness and maximum rarity,’ and ‘species richness and average rarity’. Inverse rankings were used so that the least valuable site had a ranking of 1, and the most valuable site had a ranking of 54 for each criterion and combination of criteria. Native species richness was the number of native species recorded at the site, native being defined as ‘naturally occurring in the Namoi catchment. Maximum rarity rankings were determined by sorting individual site data by the number of species recorded once, then twice, then three times etc., across all sites through to species recorded 54 times. A score could not be determined for this measure, only a ranking of sites so it is unknown whether any sites should have been ranked equally. Average rarity was computed as the average sample frequency of species occurring at the site: (Σ [frequency of occurrence across all sites of each plant species]) / (number of species recorded at the site).

Differences in species richness for various components of the vegetation and bird communities and biodiversity conservation value criteria were analysed among vegetation types and grazing intensities using parametric analysis of variance (AOV) and Tukey's pairwise comparisons where variables were normally distributed. Non-parametric Kruskal–Wallis AOV was used when variables could not be transformed to achieve normality. These analyses were completed in Statistix 8 (Analytical Software 2003). Efficiency of biodiversity conservation value measures was determined by the number of species present at least once and more than once in the 16 (top 30%) most valuable sites for plant or bird conservation, of the total 54.

Spearman correlations indicated that many of the environmental and habitat variables measured at each site were correlated and redundant. A reduced set of environmental variables (mean rainfall, time since last flood, proportions of clay, silt and sand, pH, EC, Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, ECEC, N and exchangeable K [EK]) was used to determine the impact of environmental variables on vascular plant biodiversity. Data for most soil variables were from 20–30 cm depth as there was a high correlation between the surface 0–5 cm and 20–30 cm, but both surface and subsurface depths were used for EK as the two were not correlated. For habitat variables, the number of large trees, stags, small trees, saplings, tall shrubs and pieces of CWD; the canopy, tall shrub, low shrub, herbaceous, litter and total ground cover; woody, herbaceous, litter and CWD biomass, and tree and shrub species richness were used to determine the impact of vegetation structure on bird communities.

Principal components analysis (PCA) was performed in PRIMER 6.1.11 (Clarke & Warwick 2006) on the reduced set of environmental (soil) and habitat (vegetation structure and composition) variables to decrease dimensionality, highlight variables of greatest importance in explaining plant and bird community composition, and ordinate sites according to environmental and habitat variables. Variables were first examined to ensure they were normally distributed, and standardised by subtracting the mean of the variable from each value and dividing by the standard deviation. PCA was subsequently performed on the correlation matrix (Clarke & Gorley 2006), the principal components (PCs) describing the influence of environmental variables on plant communities and habitat variables on bird communities.

Non-metric multi-dimensional scaling (NMDS) was used to ordinate sites separately according to vascular plant and bird community composition in PRIMER (Clarke & Warwick 2006). Plant cover and bird count data were log-transformed (ln[x+1]) and site similarity calculated using the Bray–Curtis statistic. Vegetation type and grazing intensity were overlaid on the NMDS ordinations to illustrate patterns in plant or bird community composition explained by these factors. Analysis of Similarities (ANOSIM) tested for significant differences in plant or bird community composition between vegetation types and grazing intensities. River red gum and coolibah communities only were used in the investigation of grazing effects as a near balanced design was achieved in these vegetation communities.

Native species richness and site maximum rarity were superimposed on the NMDS ordination in order to detect groups of species-rich and high conservation value sites. Spearman correlations determined relationships between native species richness and maximum rarity, site NMDS coordinates, each of the five PCs and each environmental or habitat variable used in PCA. Canonical correlation analysis (CAP) (Anderson & Robinson 2003) was used to illustrate the relationship between the strongest environmental gradient (PC1) and vegetation or bird community composition. CAP is a constrained analysis designed to highlight correlations between biotic data (plant cover or bird abundance) and a composite environmental, habitat or management gradient, in this case determined by PCA.

Site conservation value was illustrated in scatter plots of species richness ranks against maximum rarity ranks. The species richness and maximum rarity axes of the scatter plot were divided in half to form boundaries for high and low conservation value categories. Vegetation type and grazing intensity were overlaid on the scatter plots to illustrate their influence on biodiversity conservation value. Grazing-sensitive and grazing-tolerant indicator species in vegetation and bird communities were determined using SIMPER in PRIMER. The procedure identified species contributing most to Bray–Curtis dissimilarities between groups of sites. Species contributing 80% of the dissimilarity between groups were classed as grazing-sensitive or grazing-tolerant according to their abundance in sites subject to low grazing intensity and regular grazing, respectively. Intermittent grazing was not considered in this analysis as the focus was on extremes in grazing management intensity. The number of grazing-sensitive and grazing-tolerant plant or bird species was overlaid on an NMDS ordination of sites in order to highlight patterns in the distribution of these species according to site explanatory variables.

6.3 Results

6.3.1 Vegetation

A total of 260 plant species was recorded at 54 sites across the lower Namoi floodplain (Appendix 8). Exotic species totalled 79, several of these being listed as noxious under the *Noxious Weeds Act 1993*. No species was protected under state or federal legislation. The two most native species-rich sites were a grassland site north-east of Walgett with 53 native plant species, and a coolibah site north-west of Wee Waa also with 53 native plant species. Native species richness did not differ between vegetation types ($F_{4,49} = 1.49$, $P = 0.22$), nor did the average ranking of sites for native plant species richness by vegetation type ($F_{4,49} = 1.68$, $P = 0.17$) (Table 6.1). More exotic ($F_{4,49} = 17.0$, $P < 0.001$) and annual ($F_{4,49} = 7.89$, $P < 0.001$) species were recorded in river red gum sites than other vegetation types, although grassland sites also contained high exotic and annual species richness. The majority of native plant species recorded during vegetation survey could be considered rare, most being recorded in 1–5 sites of a total of 54 sites (Figure 6.3). Three native species, tarvine (*Boerhavia dominii*), black roly poly (*Sclerolaena muricata*) and climbing saltbush (*Einadia nutans*) were recorded at most sites. Maximum rarity ranking varied between vegetation types ($F_{4,49} = 2.85$, $P = 0.03$), river red gum sites ranking higher than coolibah and myall (Table 6.1).

Table 6.1: Mean plant species richness and conservation value rankings for each vegetation type.

	Vegetation type				
	River red gum	Coolibah	Grassland	Myall	Black box
Number of sites	12	17	15	6	4
Total species richness	48.42a	39.06b	39.93ab	40.33ab	44.5ab
Native species richness	29.58a	32.18a	28.47a	32.67a	38.75a
Exotic species richness	18.83a	6.88b	11.47ab	7.67b	5.75b
Annual species richness	19.92a	12.82b	16.00ab	12.17b	12.50b
Perennial species richness	28.50b	26.24a	23.93ab	28.17a	32.00a
Native maximum rarity ranking*	38.17a	22.00b	29.40ab	24.33b	16.50ab
Native average rarity ranking*	43.33a	17.53b	26.77ab	25.58ab	31.00ab

* Higher maximum and average rarity ranking denotes greater abundance of rare species. Letters represent significant differences between groups (Tukey's HSD) where $P \leq 0.05$.

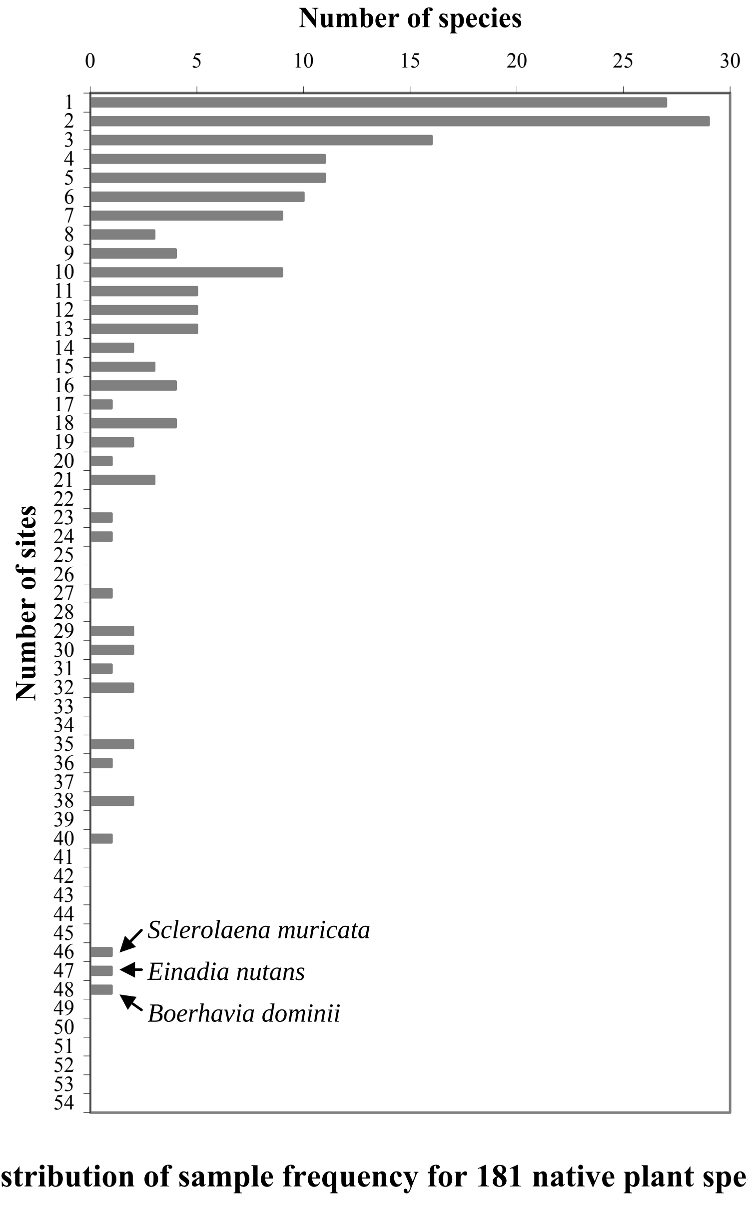


Figure 6.3: Distribution of sample frequency for 181 native plant species.

Native species richness ranks did not differ significantly with grazing intensity ($F_{2,26}$ 2.44, $P = 0.11$: Table 6.2), and nor did exotic ($F_{2,26}$ 2.32, $P = 0.12$), annual ($F_{2,26}$ 2.54, $P = 0.10$) or perennial ($F_{2,26}$ 1.35, $P = 0.28$) species richness ranks. However, maximum rarity ranks differed with grazing intensity ($F_{2,26}$ 3.76, $P = 0.04$), low grazing intensity sites having higher plant conservation value than regularly grazed sites, and intermittently grazed sites having intermediate value.

Table 6.2: Mean plant species richness and conservation value rankings for each grazing intensity.

		Grazing intensity	
	Low	Intermittent	Regular
Number of sites	9	7	13
Total species richness ranking	15.22a	17.86a	12.31a
Native species richness ranking	19.56a	12.57a	12.08a
Exotic species richness ranking	12.78a	20.43a	12.62a
Annual species richness ranking	12.67a	20.00a	12.67a
Perennial species richness ranking	17.89a	14.86a	11.85a
Maximum rarity ranking	20.56a	14.71ab	11.31b
Average rarity ranking	16.56a	17.14a	16.56a

* Inverse ranks, higher ranking denotes greater abundance of attribute. Letters represent significant differences between groups (Tukey's HSD) where $P \leq 0.05$.

Of the five criteria used to rank sites for vascular plant conservation value (Table 6.3), the maximum rarity criterion alone was the most efficient in terms of representing the greatest number of species in the top 16 sites. However the average rarity criterion resulted in the greatest number of native species represented more than once in 16 sites. There was overlap in the sites ranked highest by each criterion: 29 of the 54 sites surveyed were ranked in the top 16 sites by at least one criterion. Six sites were ranked in the top 16 sites by one criteria, eight sites were ranked by two criteria, six sites were ranked by three criterion, five sites were ranked by four criteria and four sites were ranked by all five criteria.

Table 6.3: Efficiency of five criteria for ranking sites according to vascular plant conservation value (16 highest ranked sites).

Criteria	Native species represented once (%)	 represented twice (%)
Native species rich and average rarity	84	61
Native species richness only	80	60
Average rarity only	84	64
Maximum rarity only	90	59
Max rarity and native species rich	86	62

Principal components analysis (Figure 6.4) showed several gradients in the environmental (soil and rainfall) data. The first PC explained 29.3% of the variability in the data and approximated a moisture availability gradient, while PC2 accounted for a further 27.5% of the variability and represented decreasing soil clay content (Table 6.4). The third and fourth PCs (PC3 and PC4) accounted for 13.3 and 9.3% of the variability in the environmental data and separated red ridges or prior stream formations from the clay floodplains.

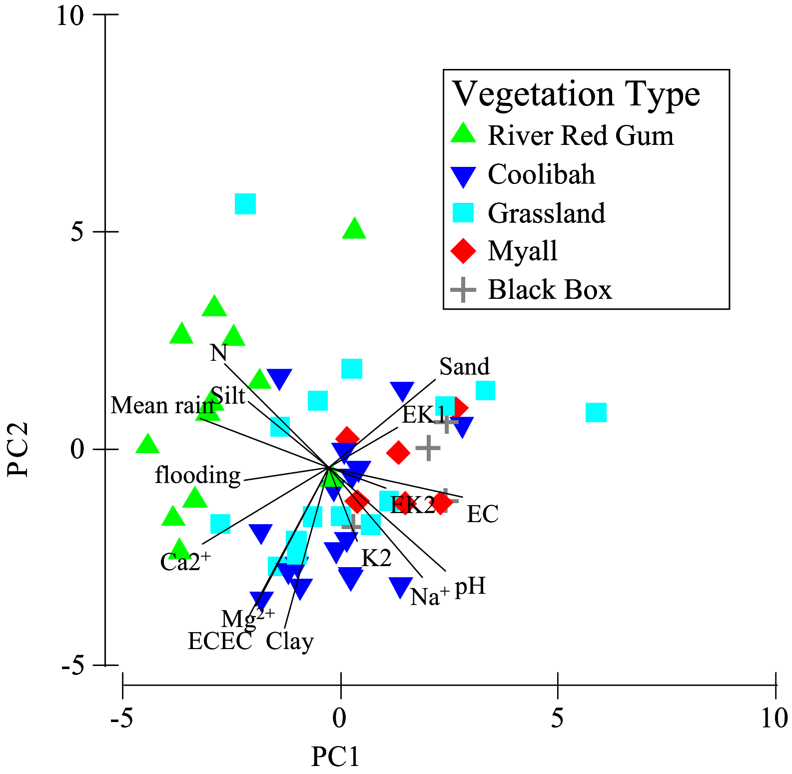


Figure 6.4: PCA ordination of environmental variables by vegetation type, showing sites in each vegetation type in relation to the first two PCs.

Table 6.4: Five principal components representing a linear combination of 15 environmental variables and the variance explained by each. Bold font indicates highest factor loadings on each PC.

	PC1	PC2	PC3	PC4	PC5
Eigenvalues	4.40	4.13	2.00	1.40	0.87
Variance (%)	29.30	27.50	13.30	9.30	5.80
Cumulative variation (%)	29.30	56.90	70.20	79.50	85.30
	Environmental Variables Factor Loadings				
Mean rainfall	**–0.354**	0.134	–0.174	–0.200	–0.177
EK$_{surrface}$	0.190	0.108	–0.073	**0.453**	**0.573**
pH	**0.325**	–0.287	–0.026	0.054	0.251
EC	**0.372**	–0.083	–0.094	0.111	**–0.428**
Ca^{2+}	**–0.347**	–0.214	0.036	0.058	**0.436**
Mg^{2+}	–0.199	**–0.386**	–0.153	–0.033	–0.196
K^{+}	0.081	–0.204	**0.615**	–0.022	–0.067
Na^{+}	0.259	**–0.307**	**–0.218**	0.084	–0.177
ECEC	–0.220	**–0.416**	–0.067	0.050	0.035
EK$_{subsurface}$	0.160	–0.058	**0.648**	–0.029	–0.069
Clay (%)	–0.120	**–0.447**	0.001	0.078	–0.007
Silt (%)	–0.221	0.180	0.160	**0.563**	–0.165
Sand (%)	0.296	0.243	–0.078	**–0.403**	0.166
Flooding	–0.233	–0.038	0.194	**–0.445**	0.199
N	–0.284	0.284	0.107	0.202	–0.169

The NMDS ordination of sites showed patterns in plant community composition as a result of both vegetation type and grazing intensity (Figure 6.5). Plant community composition differed between vegetation types (ANOSIM, $R = 0.551$, $P < 0.001$), but pairwise tests showed that grasslands and myall, and grasslands and black box were not floristically different from each other. After three derived grassland sites were re-coded as their original vegetation type (i.e. black box), vegetation type was significant (ANOSIM $R = 0.583$, $P < 0.001$), as were all pairwise comparisons. Plant community composition also differed significantly in relation to grazing intensity in river red gum (ANOSIM, $R = 0.593$, $P = 0.002$) but not coolibah (ANOSIM, $R = 0.593$, $P = 0.196$). River red gum pairwise comparisons showed that regularly and intermittently grazed sites were significantly different to those grazed regularly.

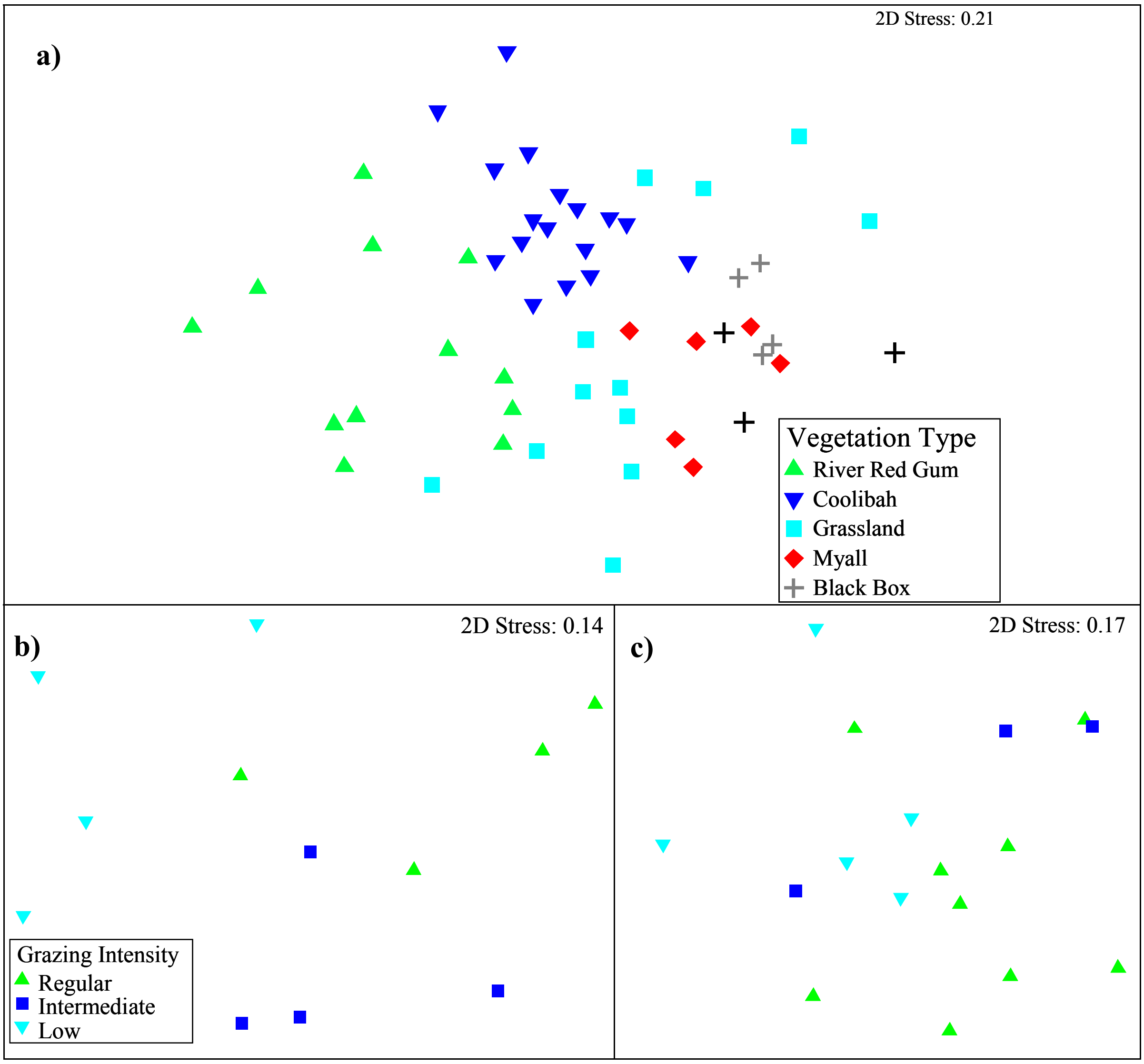


Figure 6.5: NMDS ordination of plant community composition at 54 sites by (a) vegetation type (derived grasslands are coloured black) and grazing intensity in (b) river red gum and (c) coolibah sites.

Bubble plots (Figure 6.6) showed that sites with high native species richness (Figure 6.6a) didn't necessarily have high maximum rarity ranking (Figure 6.6b). The highest ranked sites for maximum rarity were those furthest from the centre of the NMDS ordination.

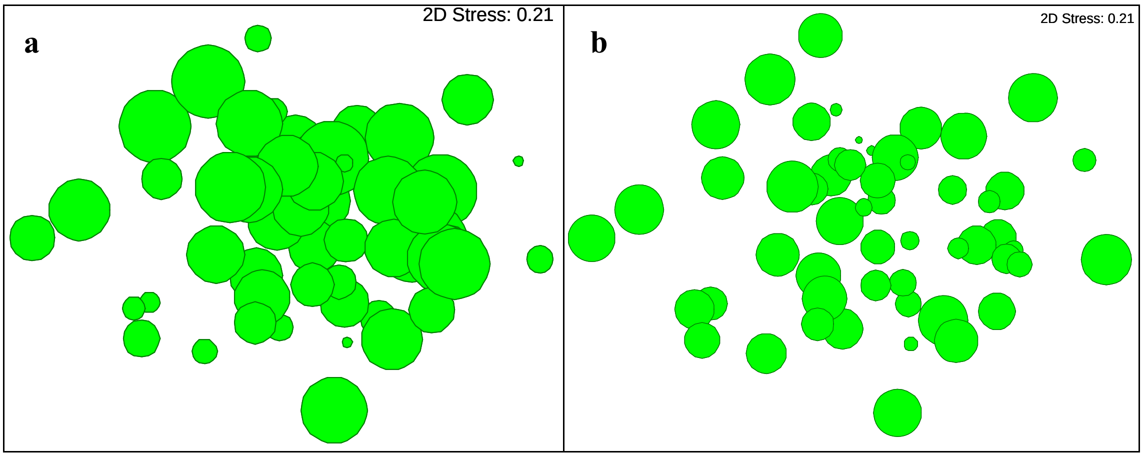


Figure 6.6: NMDS ordination of 54 sites based on plant composition, with (a) native plant species richness and (b) maximum rarity inverse rankings superimposed (larger bubbles represent higher native species richness and maximum rarity).

Native species richness and maximum rarity of sites were weakly correlated with each other (Table 6.5). Maximum rarity was correlated with the first NMDS axis and the second (PC2) and fifth (PC5) PCs. Native species richness was not correlated with either MDS axis and only showed a weak correlation with the first PC (PC1). The first NMDS axis was positively correlated with mean rainfall, flooding and N, and negatively correlated with pH, EC, Na, $EK_{surface}$ and Mg. The first NMDS axis was most strongly correlated with the first PC axis (PC1). The second NMDS axis was positively correlated with silt and N, and negatively correlated with Mg, Na, ECEC and clay. The second NMDS axis was most strongly correlated with the second PC (PC2). CAP analysis (Figure 6.7) confirmed that PC1 (representing a soil moisture gradient) had a highly significant influence on vegetation type and composition.

Table 6.5: Spearman rank correlations between native species richness ranking, maximum rarity ranking, NMDS axes and five principal components (note: a larger correlation matrix is provided in Appendix 6).

	Native rich. rank	Max. rare rank	PC1	PC2	PC3	PC4	PC5	NMDS1
Max. rare rank	0.28*							
PC1	0.29*	–0.16						
PC2	0.08	0.42**	0.06					
PC3	0.03	–0.15	0.02	–0.07				
PC4	–0.08	0.05	0.07	0.08	0.07			
PC5	–0.02	–0.43**	0.17	–0.01	–0.04	0.06		
NMDS1	–0.15	0.27*	–0.73**	0.15	0.09	–0.29*	–0.36*	
NMDS2	–0.25	0.00	–0.10	0.44**	0.05	0.20	0.04	–0.04

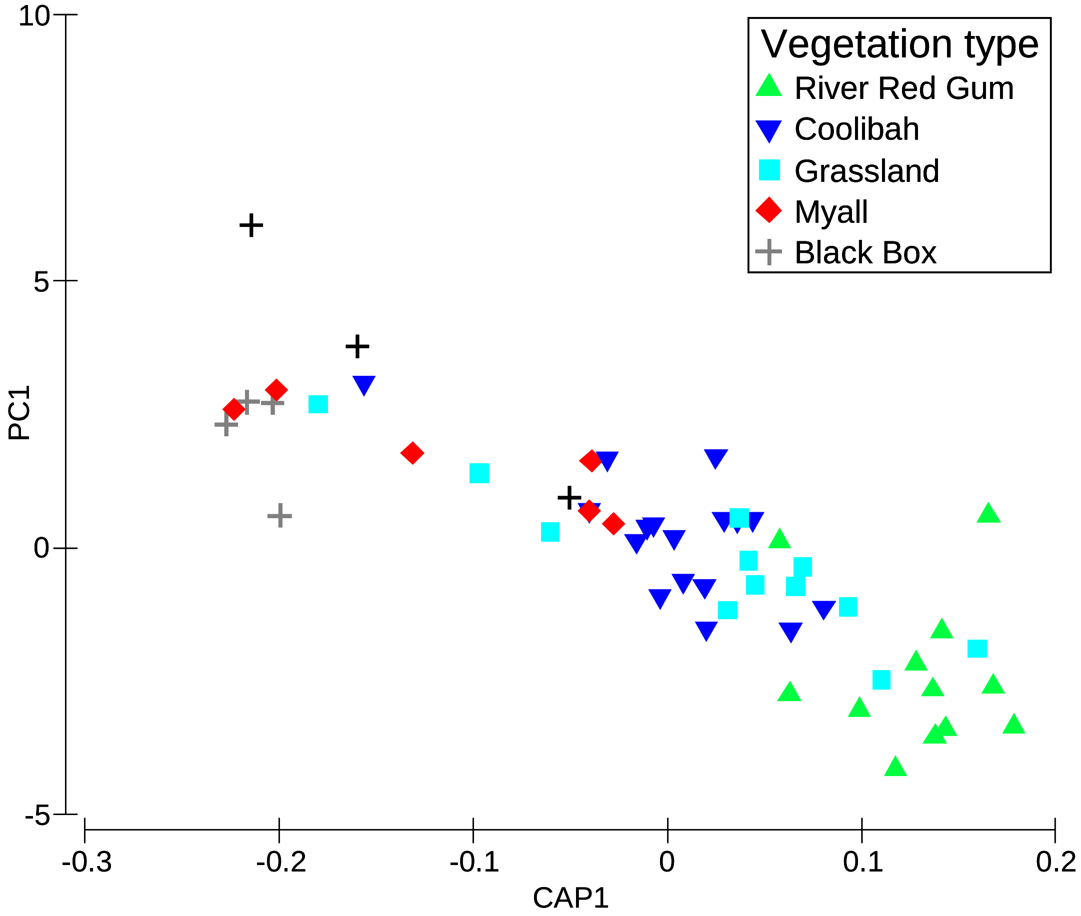


Figure 6.7: CAP analysis showing the relationship between vegetation type and vascular plant composition and a gradient of available moisture represented by PC1 ($\delta_1^2 = 0.86$, $p < 0.001$). Derived grasslands are shown in black.

A scatter plot of native species richness ranking against maximum rarity rankings (Figure 6.8) shows the distribution of sites according to conservation value. At least one site from each vegetation type was included in the top right quadrant of Figure 6.8a, representing high conservation value. Sites subjected to low grazing intensity had high maximum rarity and were more valuable for biodiversity conservation (Figure 6.8b).

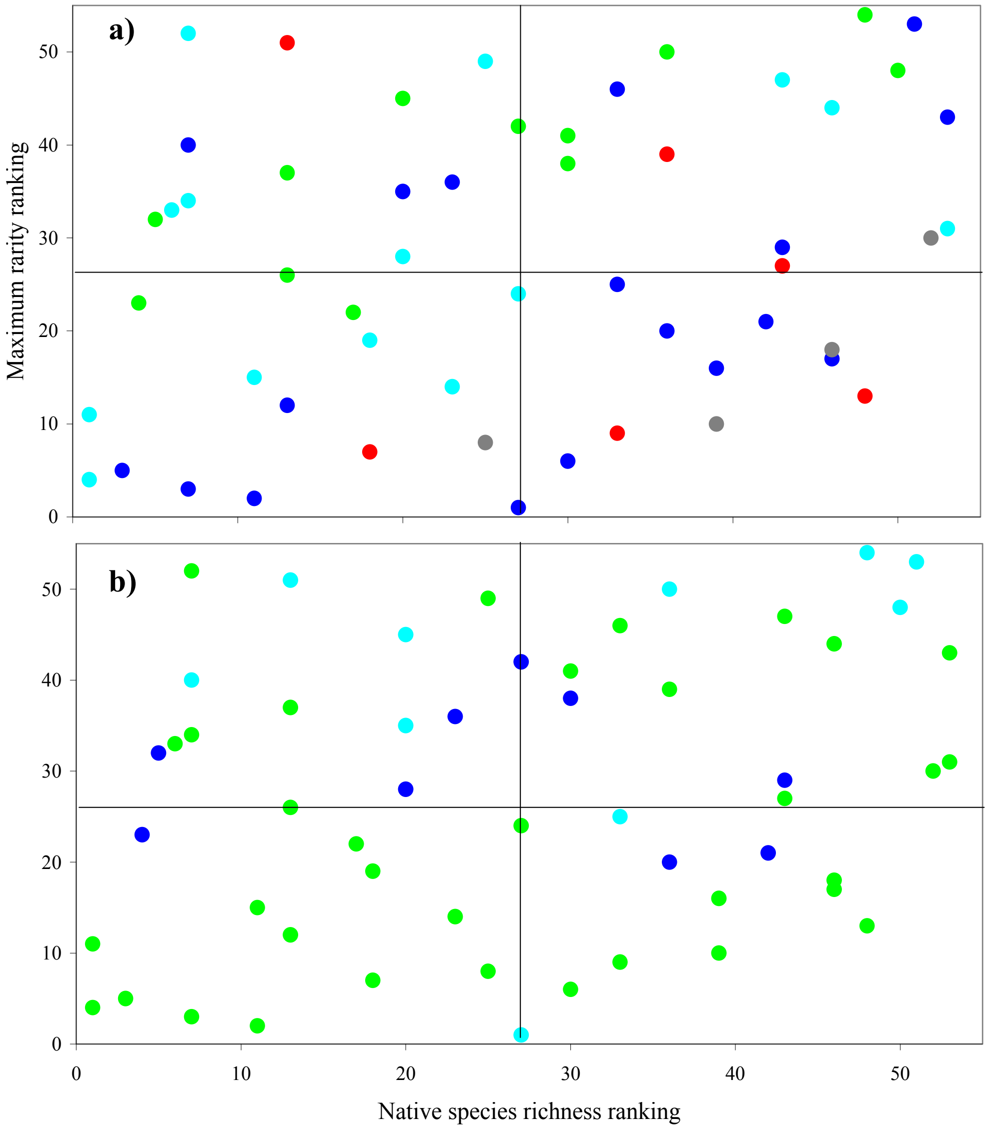


Figure 6.8: Site ranking for native species richness and maximum rarity (n = 54) by (a) vegetation type (● = river red gum, ● = coolibah, ● = myall, ● = black box, ● = grassland) and (b) grazing intensity (● = rarely grazed, ● = intermittently grazed, ● = regularly grazed). The top right quadrant shows the top 30% (16 of 54) sites used to assess efficiency of biodiversity conservation value measures.

SIMPER analysis identified 23 grazing-sensitive plant species (Table 6.6) and 21 grazing-tolerant species (Table 6.7). Grazing-sensitive species were mostly palatable, highly favoured species, the majority preferring wetter areas and of low fecundity. Grazing-tolerant species were often not palatable to sheep or cattle, or only palatable when young, and most were weedy or colonising species (Table 6.7). The NMDS plot of grazing-sensitive species shows a concentration of sites with grazing-sensitive species in the top left quadrant (Figure 6.9a). Grazing-tolerant species were more evenly distributed across the NMDS bubble plot (Figure 6.9b).

Table 6.6: Grazing-sensitive species (SIMPER), their contribution to Bray–Curtis dissimilarity and abundance at regularly (n = 13) and rarely (n = 9) grazed river red gum and coolibah sites.

		Mean abundance (cover, %)		
Species	Contribution (%)	Rarely grazed	Regularly grazed	Palatable? †
Paspalidium jubiflorum	4.39	1.67	1.46	Yes
Enteropogon acicularis	4.35	0.72	0.68	When young
Einadia nutans	4.17	1.07	0.97	Yes
Muehlenbeckia florulenta	3.75	0.80	0.38	Yes
Tetragonia tetragonioides	2.79	0.61	0.24	Yes
Sporobolus mitchellii	2.58	0.66	0.19	When young
Asperula conferta	2.49	0.63	0.09	Yes
Carex inversa	2.24	0.51	0.07	No
Eriochloa pseudoacrotricha	2.20	0.58	0.32	When young
Leptochloa divaricatissima	1.65	0.35	0.16	When young
*Bromus cartharticus**	1.32	0.23	0.14	When young
Eriochloa procera	1.26	0.31	0.00	When young
Dichanthium sericeum subsp. *sericeum*	1.08	0.28	0.00	When young
Lomandra longifolia	1.08	0.28	0.00	No
Eragrostis setifolia	1.04	0.17	0.11	When young
Cyperus lucidus	0.99	0.13	0.12	Yes
Goodenia fascicularis	0.93	0.16	0.11	Yes
Tribulus micrococcus	0.87	0.22	0.00	No
Eremophila debilis	0.84	0.19	0.00	Yes
Commelina cyanea	0.80	0.20	0.00	Yes
*Echinochloa colona**	0.80	0.13	0.10	Yes
Cynodon dactylon	0.77	0.18	0.00	Yes
*Xanthium occidentale**	0.68	0.13	0.05	No

*exotic species; † sources: Cunningham et al. (1992), Jacobs et al. (2008).

Table 6.7: Grazing-tolerant species identified by SIMPER with their contribution to Bray–Curtis dissimilarity and mean abundance at regularly (n = 13) and rarely (n = 9) grazed river red gum and coolibah sites.

		Mean abundance (cover, %)		
Species	Contribution (%)	Rarely grazed	Regularly grazed	Palatable?[†]
Sclerolaena muricata	6.16	0.81	1.67	No
*Phyla canescens**	4.07	0.55	0.56	No
Paspalidium gracile	3.06	0.09	0.80	When young
*Rapistrum rugosum**	3.02	0.19	0.82	No
*Tribulus terrestris**	2.35	0.08	0.55	No
Panicum decompositum	2.19	0.22	0.48	When young
Medicago polymorpha	2.17	0.15	0.48	Yes
Salsola kali	1.90	0.15	0.41	No
Boerhavia dominii	1.40	0.23	0.28	Yes
Phalaris paradoxa	1.26	0.00	0.35	Yes
Atriplex semibaccata	1.25	0.00	0.29	Yes
Sporobolus caroli	1.18	0.11	0.22	When young
Sclerolaena birchii	1.07	0.00	0.30	No
Sida trichopoda	1.00	0.14	0.16	When young
Eleocharis plana	0.93	0.05	0.24	When young
Dissocarpus biflorus var. *cephalocarpus*	0.84	0.00	0.18	No
Rumex brownii	0.80	0.01	0.23	No
Sclerolaena bicornis var. *horrida*	0.77	0.00	0.16	No
Marsilea drummondii	0.72	0.09	0.14	Yes
*Eragrostis cilianensis**	0.68	0.00	0.19	When young
Trianthema triquetra	0.65	0.01	0.19	No

*exotic species; [†] sources: Cunningham et al. (1992), Jacobs et al. (2008).

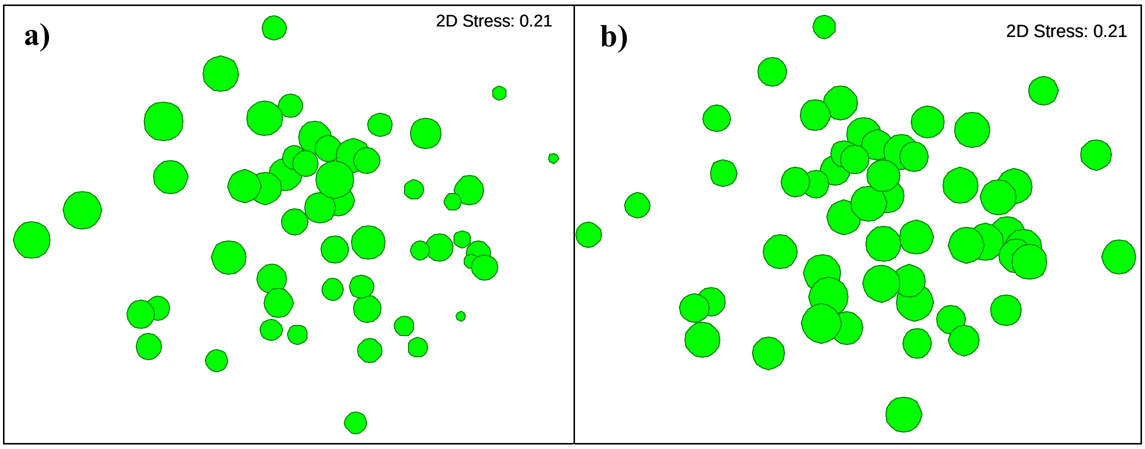


Figure 6.9: NMDS bubble plot showing number of (a) grazing-sensitive and (b) grazing-tolerant species at 54 sites throughout the study region.

6.3.2 *Birds*

Some 89 bird species were recorded during surveys at 54 sites across the lower Namoi floodplain (after waterbirds were excluded) (Appendix 9). Of the species recorded, three (brown treecreeper, grey-crowned babbler and glossy black cockatoo) are listed as vulnerable under the *Threatened Species Conservation Act 1995* (*TSC Act 1995)*, nine are listed as declining in the sheep–wheat belt of NSW (Reid 1999), 25 are decreasers in the BBS bioregion, 15 are decreasers in DRP, 26 are increasers in BBS and 39 are increasers in DRP (Barrett & Silcocks 2002). Two species (rock dove and common starling) were exotic. The most species-rich site was a river red gum-dominated site with 30 bird species north-east of Pilliga. Native species richness differed significantly between vegetation types ($F_{4,49} = 6.62$, $P < 0.001$), river red gum sites being more species-rich and grasslands less species-rich than other vegetation types (Table 6.8). There was no significant difference in average rarity ranking ($F_{4,49} = 1.91$, $P = 0.12$) or maximum rarity ranking of sites between vegetation types ($F_{4,49} = 0.63$, $P = 0.65$).

Table 6.8: Mean bird species richness and conservation value rankings for each vegetation type.

	Vegetation type				
	River red gum	Coolibah	Grassland	Myall	Black box
Number of sites	12	17	15	6	4
Native species richness	15.42a	12.88a	6.33b	11.50ab	8.25ab
Maximum rarity ranking *	29.67a	30.647a	25.33a	25.67a	18.50a
Average rarity ranking*	29.67a	29.23a	18.73a	35.67a	31.00a

* Higher maximum or average rarity ranking denotes greater abundance of rare species Letters denote significantly different groups as defined by Tukey's HSD($P \leq 0.05$).

Most native bird species recorded during bird surveys were recorded in four or fewer sites across the region (Figure 6.10). The three most frequently recorded species were galah, crested pigeon and Australian magpie. Of the five criteria used to assess conservation value, 'maximum rarity and species richness' was the most efficient, but the ranking of sites according to 'species richness only' resulted in more species being represented at multiple sites (Table 6.9).

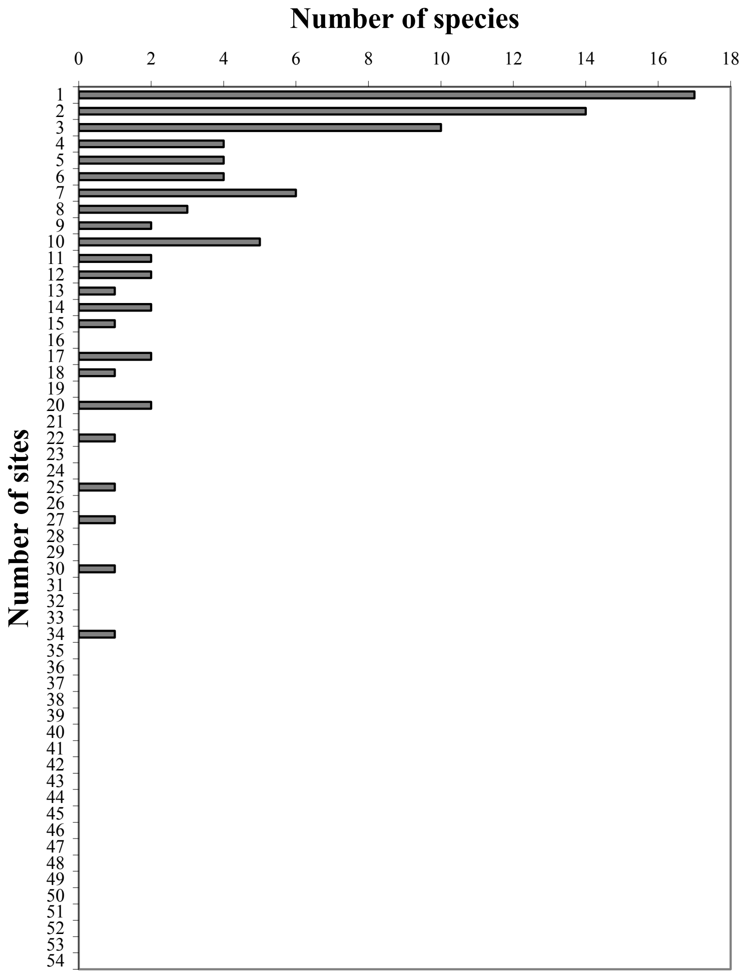


Figure 6.10: Distribution of sample frequency for 87 bird species.

Table 6.9: Efficiency of five criteria for ranking sites according to bird conservation value in the 16 highest ranked sites (top 30%) according to each criterion.

Criteria	Species represented once (%)	Species represented twice (%)
Native species richness and unique	87	64
Native species richness only	91	67
Unique only	86	59
Maximum rarity	91	51
Maximum rarity and native species richness	92	66

Native species richness differed significantly with grazing intensity in river red gum and coolibah sites ($F_{2,26} = 9.55$, $P < 0.001$), sites with low grazing intensity having higher native species richness than sites that were regularly or intermittently grazed (Table 6.10). Maximum rarity ($F_{2,26} = 5.70$, $P = 0.009$) and average rarity ($F_{2,26} = 7.27$, $P = 0.003$) also differed between grazing intensities, sites with low grazing intensity having higher maximum and average rarity than sites that were regularly or intermittently grazed.

Table 6.10: Mean bird species richness and conservation value rankings for each grazing intensity.

	Grazing intensity Low	 Intermittent	 Regular
Number of sites	9	7	13
Native species richness ranking	19.56a	12.57a	12.08a
Maximum rarity ranking	21.78a	13.29ab	11.23b
Average rarity ranking	42.44a	30.29ab	19.92b

* Inverse rank; higher ranking denotes greater abundance of attribute. Letters denote significantly different groups as defined by Tukey's HSD($P \leq 0.05$).

Principal components analysis (Figure 6.11) identified five statistically independent components that accounted for 78% of the total variance in the habitat complexity data (Table 6.11). Each PC represented a linear combination of the original 18 habitat variables. Bold font indicates factors with higher loading and potentially, the most influential factors on the PC.

The first PC axis represented decreasing numbers of small trees and some of the variables determined by it such as declining canopy cover, number of pieces of CWD and above-ground live woody biomass. The second PC represented decreasing ground cover components and increasing numbers of tall shrubs and tall shrub richness. The third PC represented increasing number of tall shrubs, tree species richness, and herbaceous ground cover and biomass, and decreasing litter biomass. The fourth PC represented increasing number of large trees, CWD biomass and tree species richness, and decreasing number of saplings and tall shrub cover. The last PC (PC5) represented decreasing low shrub cover and number of tall shrubs, and increasing tree species richness. These PCs were useful in describing the influence of habitat characteristics on bird communities.

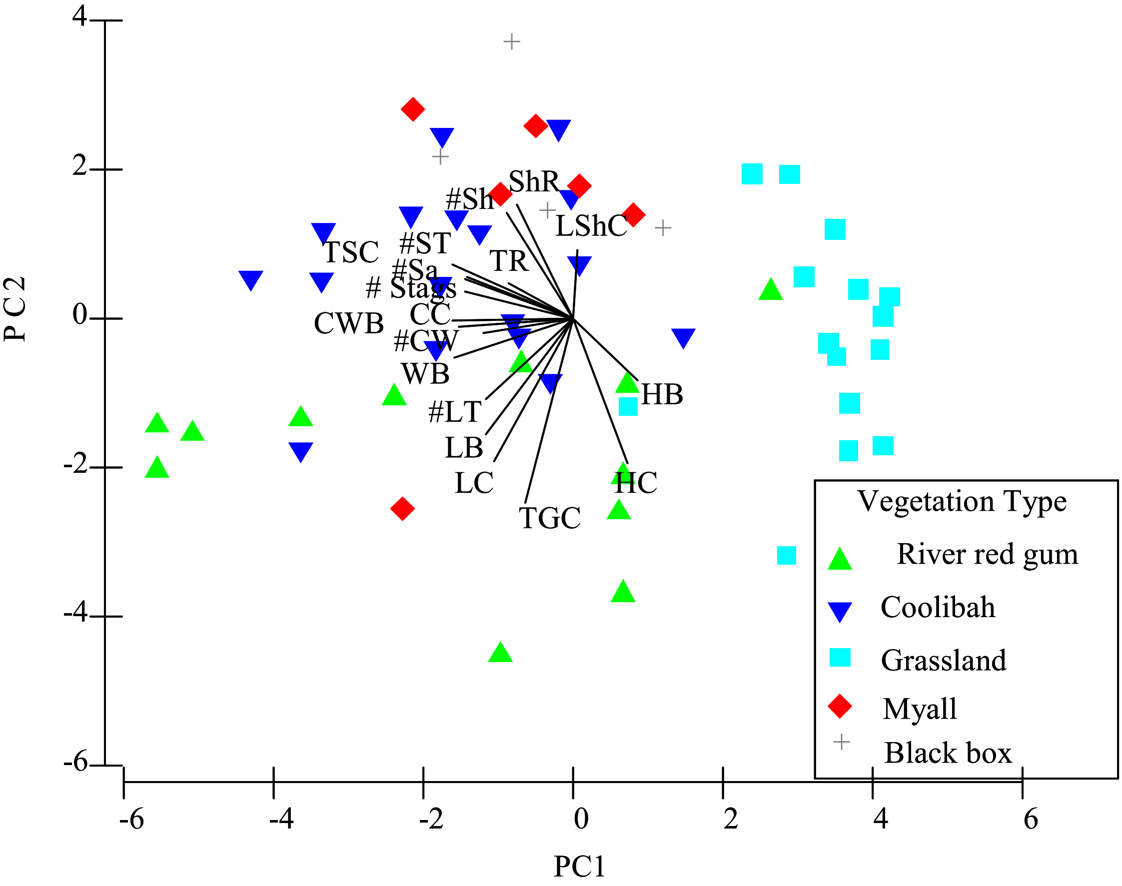


Figure 6.11: PCA ordination of habitat characteristics by vegetation type on the first two PCs. Habitat variable codes are listed in Table 6.11

Table 6.11: Five principal components representing linear combinations of 18 habitat characteristics and the variance explained by each.

		PC1	PC2	PC3	PC4	PC5
Eigenvalues		7.34	3.08	1.40	1.15	1.05
Variance (%)		40.80	17.10	7.80	6.40	5.90
Cumulative variation (%)		40.80	57.90	65.70	72.10	77.90
	Code	Habitat Characteristics Factor Loadings				
No. large trees[1]	#LT	–0.23	–0.22	0.25	**0.42**	0.01
No. stags[1]	#Stags	–0.29	0.07	–0.19	–0.01	0.04
No. small trees[1]	#ST	**–0.32**	0.14	0.02	–0.12	0.03
No. saplings[1]	#Sa	–0.29	0.10	0.14	**–0.38**	0.01
No. tall shrubs[1]	#Sh	–0.18	0.28	**0.31**	–0.22	**–0.35**
No. CWD pieces[1]	#CW	**–0.30**	–0.02	0.00	**0.32**	–0.01
Canopy cover	CC	**–0.32**	–0.01	0.14	–0.12	0.15
Low shrub cover	LShC	0.01	0.18	–0.01	0.24	**–0.80**
Tall shrub cover	TSC	–0.28	0.11	0.19	**–0.40**	0.03
Herbaceous cover	HC	0.15	**–0.39**	**0.39**	–0.02	–0.08
Litter cover	LC	–0.21	**–0.39**	–0.23	–0.10	–0.15
Total ground cover	TGC	–0.13	**–0.50**	0.03	–0.10	–0.16
Log(woody biomass)	WB	**–0.31**	–0.11	0.17	0.28	–0.02
Herbaceous biomass	HB	0.18	–0.17	**0.53**	–0.17	–0.14
Litter biomass	LB	–0.23	**–0.31**	**–0.30**	–0.13	–0.09
CWD biomass	CWB	–0.24	–0.04	–0.17	0.03	–0.11
Tree species richness	TR	–0.17	0.09	**0.32**	**0.33**	**0.33**
Tall shrub species richness	SR	–0.15	**0.30**	0.01	0.19	–0.06

[1] indicates log-transformation (log[x+1]).

The NMDS ordination of sites showed patterns in bird community composition due to both vegetation type and grazing intensity (Figure 6.12). ANOSIM confirmed a significant difference in bird community composition between vegetation types (R = 0.227, P < 0.001) due to differences between river red gum-dominated sites and all other vegetation types, and between coolibah and both black box and grassland ecosystems. Bird community composition also differed significantly in relation to grazing intensity in river red gum (ANOSIM, R = 0.793, P = 0.001) but not coolibah (ANOSIM, R = 0.130, P = 0.133). River red gum pairwise comparisons showed all grazing intensities were significantly different from each other in terms of composition.

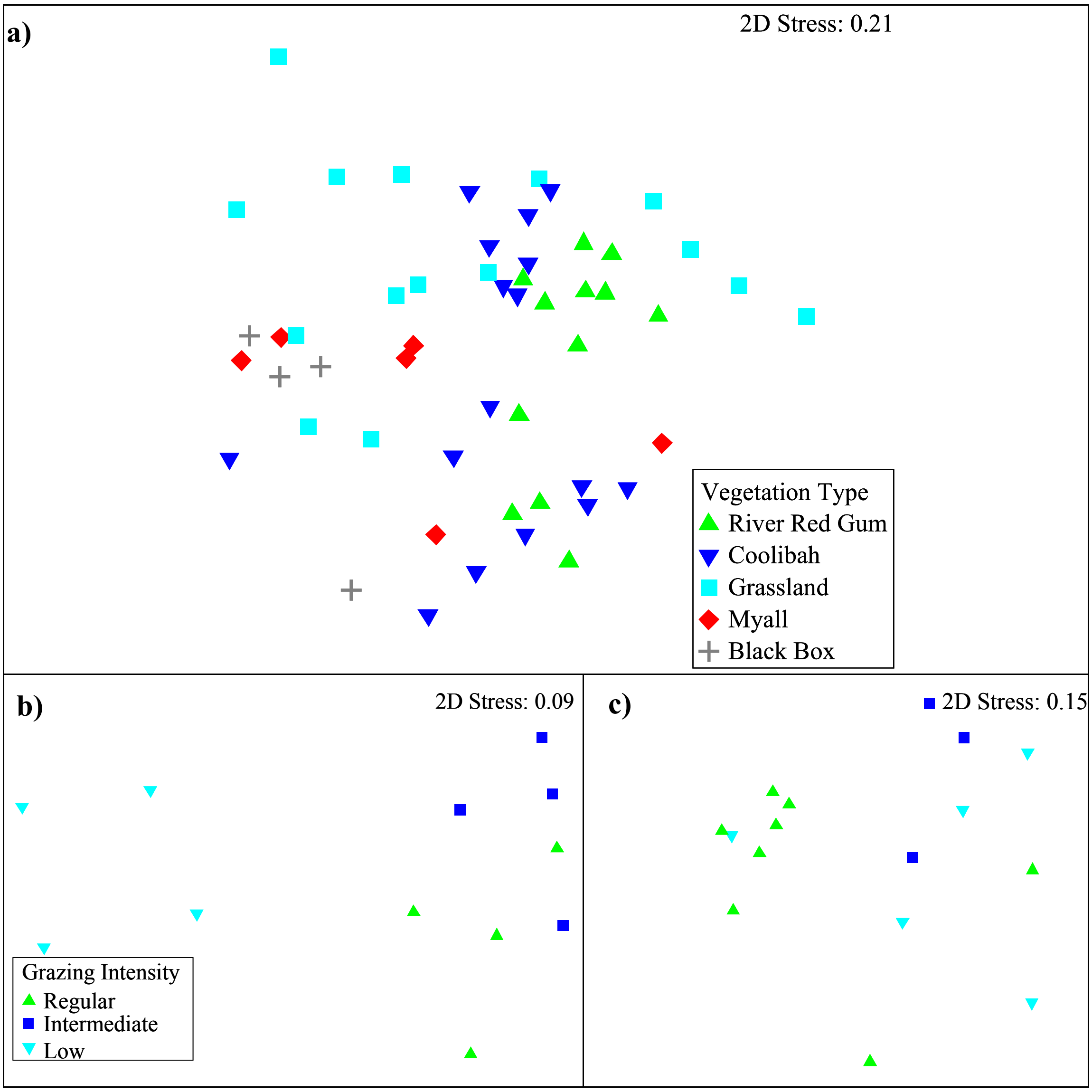


Figure 6.12: NMDS ordinations of (a) bird community composition at 54 sites by vegetation type, (b) 12 river red gum sites by grazing intensity and (c) 16 coolibah sites by grazing intensity.

Species richness and site maximum rarity were overlaid on the NMDS of bird communities (Figure 6.13). Species-rich sites had the greatest bird community average rarity and these sites grouped together in ordination space.

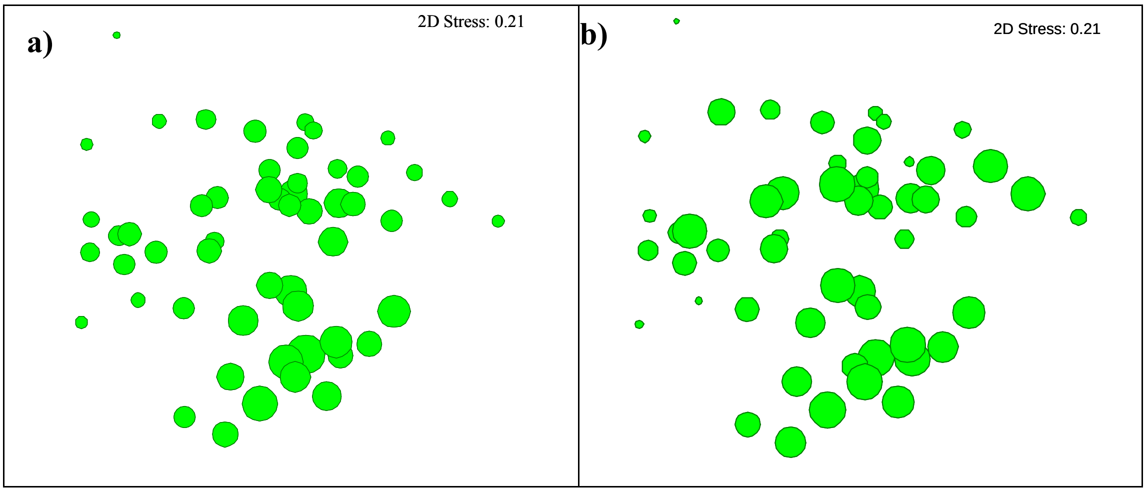


Figure 6.13: NMDS ordination of bird assemblages at 54 sites showing (a) bird species richness and (b) site maximum rarity ranking.

Native species richness and maximum rarity of the native bird data were strongly correlated with each other (Table 6.12). Maximum rarity was correlated with the first and second NMDS axes and PC1. Native species richness was correlated with the second NMDS axis and the PC2. The first NMDS axis was positively correlated with litter cover, total ground cover, woody biomass, litter biomass and number of large trees. The first NMDS axis was most strongly correlated with the second PC axis (PC2). The second NMDS axis was positively correlated with herbaceous cover and negatively correlated with cover of the canopy and tall shrubs, biomass of woody vegetation, litter and CWD, the number of small trees, saplings, shrubs, stags and pieces of CWD, and shrub species richness. The second NMDS axis was most strongly correlated with the first PC (PC1). Canonical analysis of principal coordinates (CAP analysis) confirmed that there was a strongly significant relationship between bird composition and the first PC axis (PC1) representing canopy cover (Figure 6.14).

Table 6.12: Spearman rank correlations between native species richness ranking, maximum rarity ranking, NMDS axes and five principal components (note: a larger correlation matrix is provided in Appendix 7).

	Native rich. rank	Max. rare rank	PC1	PC2	PC3	PC4	PC5	NMDS1
Max. rare rank	0.70**							
PC1	–0.69**	–0.35*						
PC2	–0.13	–0.21	–0.03					
PC3	0.10	–0.12	–0.04	–0.07				
PC4	–0.08	–0.21	–0.01	0.05	–0.05			
PC5	–0.08	–0.02	0.01	0.15	–0.10	0.08		
NMDS1	0.24	0.28*	–0.17	–0.55**	–0.07	0.00	–0.05	
NMDS2	–0.61**	–0.42**	0.56**	–0.21	–0.06	0.20	0.13	0.06

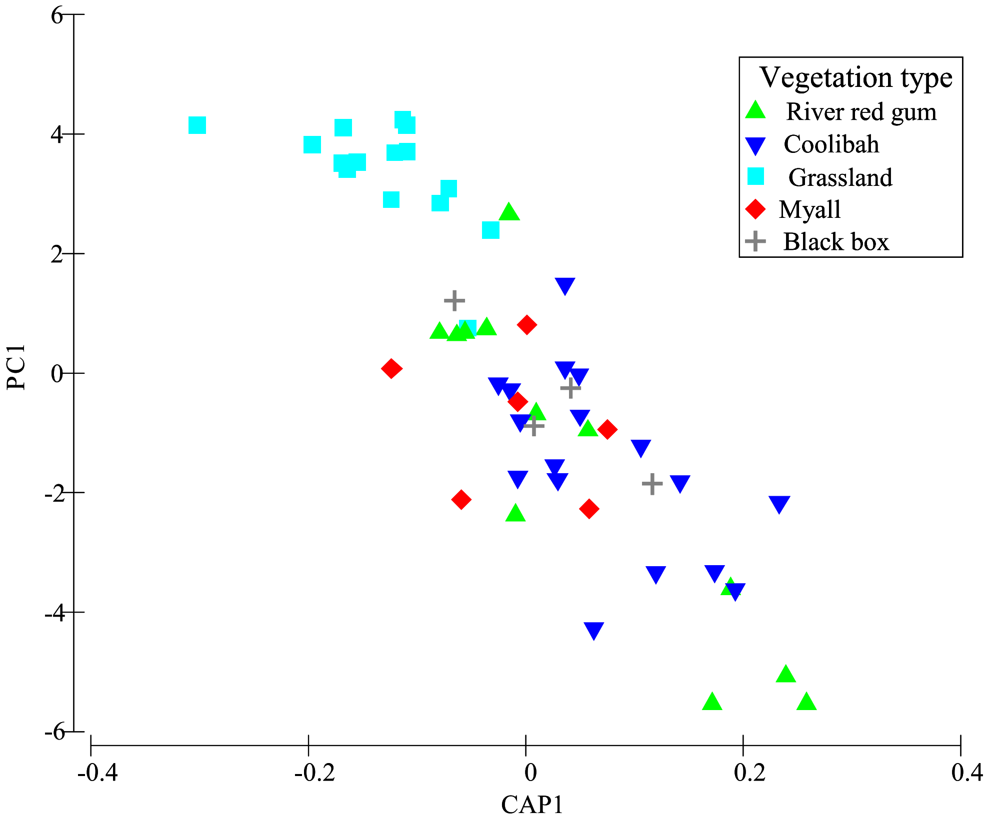


Figure 6.14: CAP analysis of bird composition showing the relationship between vegetation type and PC1, which represents a gradient in canopy cover and related variables ($\delta_1^2 = 0.76$, $p = 0.001$).

A scatter plot of species richness against maximum rarity of bird species at 54 sites (Figure 6.15) showed that the 16 sites of highest conservation value were mainly coolibah or river red gum sites in riparian zones, although two myall sites were also of high conservation importance. No grassland or black box sites featured in the high conservation value quadrant. All but two sites experiencing low grazing pressure occurred in the quadrant representing high conservation value. Sites with intermediate grazing pressure were spread throughout the plot as were sites of greater grazing pressure, though there was a concentration of regularly grazed sites in the low conservation value quadrant.

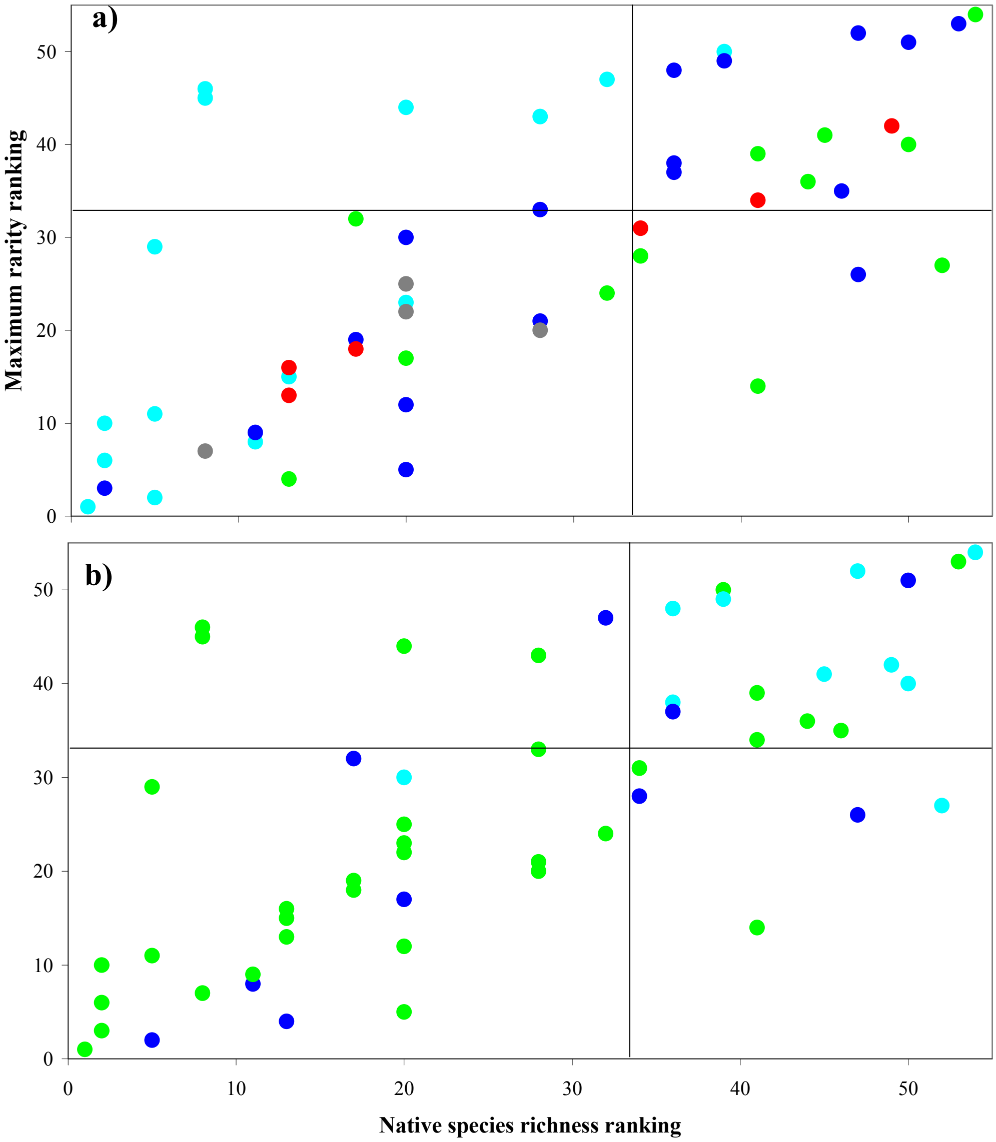


Figure 6.15: Site ranking for native species richness and maximum rarity (n = 54) by (a) vegetation type: (● = river red gum, ● = coolibah, ● = myall, ● = black box, ● = grassland) and (b) grazing intensity (● = rarely grazed, ● = intermittently grazed, ● = regularly grazed). The top right quadrant shows the top 30% (16 of 54) sites used to assess efficiency of biodiversity conservation value measures.

SIMPER identified 22 grazing-sensitive bird species (Table 6.13). Grazing-sensitive species were mainly ground or canopy feeding insectivorous woodland specialists. Three species were declining woodland birds (Reid 1999) but the majority had stable or increasing populations in the BBS and DRP bioregions (Barrett & Silcocks 2002).

Table 6.13: Grazing-sensitive species (SIMPER), their contribution to Bray–Curtis dissimilarity and abundance at regularly (n = 13) and rarely (n = 9) grazed river red gum and coolibah sites.

Species	Contribution (%)	Mean abundance (number of birds per site) Regularly grazed sites	 Rarely grazed sites
White-plumed honeyeater	5.60	0.30	1.35
Little friarbird	4.01	0.30	1.03
Red-rumped parrot	3.70	0.24	0.92
Weebill	2.64	0.23	0.51
White-browed woodswallow	2.53	0.00	0.61
Peaceful dove	2.52	0.03	0.62
White-winged chough	2.36	0.17	0.53
Australian raven	2.15	0.31	0.35
Willie wagtail	2.03	0.12	0.45
Magpie-lark	2.01	0.12	0.48
Superb fairy-wren	1.98	0.10	0.37
Variegated fairy-wren	1.79	0.07	0.31
Grey shrike-thrush	1.70	0.06	0.42
Grey fantail	1.68	0.15	0.33
Yellow-throated miner	1.51	0.14	0.18
Brown treecreeper	1.49	0.10	0.31
Sacred kingfisher	1.44	0.05	0.30
Rufous songlark	1.30	0.00	0.33
Jacky winter	1.30	0.15	0.17
Tree martin	1.14	0.05	0.27
White-winged fairy-wren	1.05	0.10	0.15
Yellow thornbill	0.99	0.00	0.22

Of the 12 grazing-tolerant bird species (Table 6.14), most were omnivorous, preferred open woodland habitats and were hollow nesters. Grazing-sensitive species were concentrated in sites in the bottom-right quadrant of the NMDS ordination (Figure 6.16a). The number of grazing-tolerant species per site was higher towards the top of the plot (Figure 6.16b).

Table 6.14: Grazing-tolerant species (SIMPER), their contribution to Bray–Curtis dissimilarity and abundance at regularly (n = 13) and rarely (n = 9) grazed river red gum and coolibah sites.

		Mean abundance (number of birds per site)	
Species	Contribution (%)	Regularly grazed sites	Rarely grazed sites
Apostle bird	1.48	0.18	0.14
Grey butcherbird	1.00	0.18	0.09
Pied butcherbird	1.14	0.19	0.09
Sulphur-crested cockatoo	3.61	0.66	0.51
Australian ringneck	2.19	0.38	0.14
Australian magpie	2.64	0.51	0.26
Crested pigeon	2.21	0.66	0.40
Eastern rosella	1.91	0.46	0.05
Cockatiel	3.13	0.75	0.32
Galah	5.39	1.43	0.9
Common starling	3.07	0.79	0.00
Noisy miner	5.49	1.23	0.40

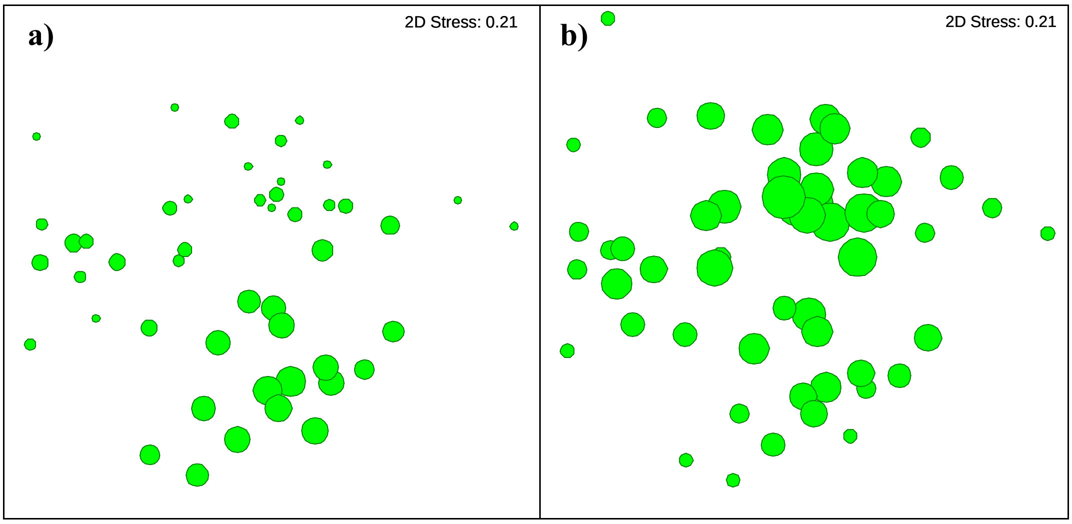


Figure 6.16: NMDS bubble plot showing number of (a) grazing-sensitive and (b) grazing-tolerant bird species present at different sites.

6.4 Discussion

6.4.1 Efficiency of Different Measures of Biodiversity Conservation Value

Some measures of biodiversity conservation were more efficient than others in ranking sites according to conservation value; although this depended on the taxon and particular conservation goal. Other studies (e.g. Pressey & Nicholls 1989) have reported similar findings. In the plant community data set, the greatest efficiency was achieved using a ranking of site maximum rarity where the goal was to represent the greatest number of species within a set number of sites. Where the goal was to represent the greatest number of species at a minimum of two separate locations, the average rarity ranking was more efficient than other criteria. A similar result was returned in the bird community data, where the combination of maximum rarity and species richness gave the greatest number of species represented at one site only, but species richness only resulted in the greatest number of species represented at a minimum of two sites. Pressey and Nicholls (1989) advocated the use of iterative procedures over criteria such as those used in the current study for prioritising sites for acquisition and inclusion in the conservation estate. The aim of the current study was to determine the relative conservation value of individual sites, however, to help landholders understand the importance of different native vegetation remnants on their properties and the impacts of grazing management on biodiversity conservation value.

6.4.2 Vegetation Type and Environmental Influences on Plant Conservation Value

Two sites were ranked equally high for native vascular plant species richness, one a grassland site near Walgett, the other a coolibah site north of Burren Junction. The grassland site was previously a coolibah, black box and belah (*Casuarina cristata*) woodland which had been cleared, and prior to the first sampling period, had been heavily grazed by sheep during a long period of below average rainfall. The site showed a high degree of microtopography (gilgai) and therefore provided a range of soil moisture and physico-chemical conditions at small spatial scale. Different plant assemblages were observed in the depressions and mounds of the gilgais, more grass species occurring in the depressions and chenopod species on the mounds. Herbaceous species common to both woodland and grassland sites, including both grazing-tolerant and grazing-sensitive species, were present at this site. It is possible that the woodland species will disappear with time if the community shifts towards grassland, resulting in an overall decline in species richness. The coolibah site was characterised by variation in soil properties across a relatively small area adjacent to the site. At one end of the paddock there was a sandy red ridge grading into a duplex soil dominated by cotton bush (*Maireana aphylla*), grading into the grey clay of the coolibah-dominated area. This sequence of soil changes was associated with a prior stream formation and it may be that the surface soil at the site (i.e. grey clay) did not reflect the soil type at depth, thus creating a combination of soil properties and variety of niches for different plants to co-exist. The complexity of soil properties at this site, and the corresponding differences in flood frequency (the grey clay being flooded relatively frequently but the red ridge not) likely explain the high plant species richness at this site.

Maximum rarity of native vascular plants was recorded at a river red gum site on the Namoi River near Walgett. The species present at this site included several recorded only once. This site was the only site sampled on the Namoi River near Walgett that was dominated by river red gums; the rest were coolibah-dominated. The closest river red gum site included in the sample was near Pilliga, 100 km upstream. This site was unusual among river red gum sites in that it had only eight exotic species compared to the average of 18. It is possible that many of the exotic species in other river red gum sites had dispersed downstream from higher in the catchment but had not reached the end of the catchment yet. Highly competitive weeds such as lippia were not present at this site, but it is likely only a matter of time before this species colonises the lower catchment (Earl 2003).

Native plant richness did not differ significantly between vegetation types, but myall and black box sites had high species counts. These sites have sandier soils than the coolibah and grassland sites which both had lower species richness and average rarity. In comparison to the soils of the other vegetation types, myall and black box soils were less fertile (in terms of N) and slightly sodic, often with crusted soil surfaces. The more stressful plant–soil environment of the myall and black box soils probably helped to facilitate plant species co-existence by suppressing the dominance of highly competitive species (Huston 1979).

Total species richness was significantly higher in river red gum sites, but this was driven by substantially higher numbers of exotic species. River red gum sites also contained more annual species than other vegetation types. The number of exotic species may detract from the value of this vegetation type for biodiversity conservation, especially where exotic species cover is greater than native species cover. Weeds can have positive or negative influences on the perceived value of a site depending on their economic importance and contribution to ecosystem service provision, impact on biodiversity and ecosystems, and their effect on cultural identity (García-Llorente et al. 2008). From a conservation perspective, exotic species alter species composition through competition and predation, hybridisation with indigenous species (Pimental et al. 2001), and changes to disturbance regimes and microclimate (Mack & D’Antonio 1998).

Myall, coolibah and black box sites contained the least exotic species. In the case of myall and black box, this is likely due to the reduced flooding of these sites. Both myall and black box dominated vegetation occur on slightly elevated alluvial or aeolian deposits across the floodplain often above the flood line. Floodwaters spread weed propagules (Johansson et al. 1996), and creek lines are commonly impacted by exotic species (Hill et al. 2005). Lippia, for example, was recorded across the study area in low lying, frequently flooded vegetation (i.e. river red gum, coolibah and grasslands) but not in myall or black box. Floodwaters disperse the seed and vegetative fragments of this species (Crawford 2008; Earl 2003; Taylor & Ganf 2005). The lower fertility and sodic nature of myall and black box soils may also explain the reduced frequency of introduced species (Prober et al. 2002; Trémont & McIntyre 1994; Wijesuriya & Hocking 1999).

The longest axis of variation in floristic composition of the remnant vegetation across the lower floodplain was due to moisture availability represented by flood frequency, annual rainfall and soil EC. The most water limited sites had negative loadings on CAP1 (Figure 6.7). The majority of these sites were located near Walgett at the low rainfall end of the region. Black box dominated sites were particularly well represented, along with myall dominated sites, which also occurred on non-saline, sodic soils with a high EC and Na^+ content. Most sodic soils have reduced water holding capacity as a result of reduced porosity and the formation of impenetrable soil crusts which impede infiltration (McKenzie et al. 2004). In addition, soils with high EC have a high osmotic potential, rendering soil water less available to plants (McKenzie et al. 2004). Both sodic soils and soils with high EC values were characterised by a low shrub layer dominated by chenopods (*Atriplex*, *Sclerolaena*, *Maireana* and *Dissocarpus* species) as opposed to grasses and herbs, forming distinct plant assemblages.

Grasslands occurred naturally across the majority of the lower Namoi floodplain, but many of them have been transformed for cropping, and many of the grasslands existing now are derived from woodland as a result of extensive ringbarking to increase biomass production for grazing during the 1880s (Ferry 1978; Maher 1995). Unfortunately, records of the extent and intensity of ringbarking are not available, but the species composition of some of the grassland sites provided clues to their derivation. Three grassland sites occurring in the lower right quadrant of the NMDS (Figure 6.5a) were probably derived from black box woodlands, since they were surrounded by black box woodland and large dead branches or roots were observed at two of the sites. Two grassland sites in the top centre of the NMDS plot both contained scattered coolibah trees but may once have been denser woodland than the open, savanna structure present today. In contrast, two Mitchell grass-dominated grassland sites in the top right of the NMDS plot occurred on rarely flooded brown clay soils, and no indication of ringbarking or other clearing activities was observed at these sites.

In terms of conservation value, sites representing all of the vegetation types surveyed featured in the ‘high plant conservation value’ quadrant in Figure 6.8. A basic tenant of biodiversity conservation is that a mosaic of ecosystem types at the landscape scale is optimal, because different types of habitat in time and space are required by different species (Lambeck & Saunders 1993). Landholders that conserve the best sites on their properties will contribute to regional conservation goals while maintaining a habitat mosaic and maximum biodiversity across the landscape and potentially, a high degree of ecosystem service provision—a win–win for local agricultural industries and society in general.

6.4.3 Grazing Impacts on Site Value for Plant Conservation

Most if not all areas of the lower Namoi floodplain have experienced grazing by domestic livestock since European settlement in the 1830s. The vegetation of the study area is therefore likely to have changed over time in terms of species composition, although grazing intensity and ecosystem resilience (especially in terms of soil characteristics) vary across the landscape (Wilson 1990). Species composition and maximum rarity varied significantly with grazing intensity, and grazing-sensitive and grazing-tolerant species were distinguished. There were no differences in total, native, exotic or annual species richness between grazing intensities. Landsberg et al. (2003) and McIntyre and Lavorel (1994) found reductions in the number of rare species with increased grazing pressure, indicating a loss of heterogeneity. This was also observed in the current study as maximum rarity differed significantly with grazing intensity. Lewis et al. (2009) found that grassland communities of the Moree Plains were resilient to long-term grazing-induced change, and concluded that season and rainfall were more important determinants of community composition. Lewis et al. (2009) determined only the effects of recent grazing intensity, but acknowledged that long-term grazing may have had an influence on present grassland composition and that continued intense grazing for long periods (overgrazing) could result in the loss of dominant grasses such as *Astrebla lappacea, A. elymoides, Dichanthium sericeum, Eulalia aurea Themeda avenacea* and several other grazing-sensitive species. While moisture availability was highlighted in the current study as an important determinant of vegetation community composition, the present study demonstrates that rare species are being lost from the vegetation community as a result of livestock grazing. The lack of a significant result in the Lewis et al. (2009) study may be a product of changes in plant community composition early in the history of settlement with little recovery to present (Lunt 1997; Lunt et al. 2007). A critical difference between the study by Lewis et al. (2009) and the current study was that they examined only one vegetation type (i.e. the grasslands of the Moree Plains), whereas several different vegetation types were considered here. There are two potential explanations for the different conclusions between the Moree Plains grasslands and the current study. The first possibility is that the Moree plains grasslands are more resilient to grazing than the vegetation types examined in the current study. The Moree Plains grasslands receive greater annual rainfall than the study sites of the lower Namoi floodplain, and therefore the interaction between grazing pressure and climatic conditions may be less severe. The second possibility is that grazing impacts in the lower Namoi floodplain have been more marked as a result of longer grazing history in the area. The current study indicates that at the landscape scale, grazing may be significantly impacting biodiversity conservation value of the vegetation.

Grazing-tolerant chenopod species such as black roly poly, goat-head burr (*S. bicornis* var. *horrida*), galvanised burr (*S. birchii*) and many-horned saltbush (*Dissocarpus biflorus* var. *cephalocarpus*) are often recorded in degraded pastures as mature plants are unpalatable to grazing animals due to the presence of thorns, the seeds are easily dispersed, and they are early successional species (Beadle 1948). Of the forb species observed, soft roly poly (*Salsola kali*), tarvine and caltrop (*Tribulus terrestris*) are widespread, in high abundance, and indicate inappropriate grazing management (Beadle 1981). Tarvine and caltrop, as well as weedy saltbushes (e.g. *Atriplex semibaccata*) listed as grazing-tolerant have taproot or underground storage structures, or woody root systems that allow them to persist under heavy grazing pressure (Beadle 1948; Cunningham et al. 1992). Grasses such as slender panic (*Paspalidium gracile*) are less palatable than other grass species and only grazed when they are very young (Cunningham et al. 1992). Native millet (*Panicum decompositum*) and fairy grass (*Sporobolus caroli*) have wind dispersed seed or seed heads that can be blown long distances under the right conditions, and are early successional species, quickly colonising disturbed or drought affected grazing lands (Beadle 1948; Cunningham et al. 1992). Many introduced species including wild turnip (*Rapistrum rugosum*), clover (*Medicago polymorpha*) and stink grass (*Eragrostis cilianensis*) were also grazing tolerant. Again, these are likely to be early successional species and are often recorded in degraded pastures (Beadle 1948; Lewis et al. 2009). The prevalence and dominance of allelopathic weeds such as turnip weed and nitrogen fixing legumes (such as clover) in the study region is a concern for vascular plant conservation, as these plants change soil mycorrhizal communities (Annapurna et al. 1996) and increase soil nutrient status, facilitating the invasion of other introduced species (Prober et al. 2002).

The list of grazing-sensitive species identified in this study was counfounded by vegetation type. Species such as couch grass (*Cynodon dactylon*), rat's tail couch (*Sporobolus mitchellii*) and knob sedge (*Carex inversa*), for example, were recorded in wetter, more densely vegetated river red gum sites often experiencing less grazing pressure. Introduced species such as prairie grass (*Bromus catharticus*), awnless barnyard grass (*Echinochloa colona*) and noogoora burr (*Xanthium occidentale*) were also most commonly observed in river red gum sites as a result of dispersal by water and high nutrient availability. The river red gum sites along the Namoi River likely endured higher degrees of disturbance early in European settlement as the river was used to water stock before the discovery of subsurface water in the 1860s (Ferry 1978). The Namoi River was also used to drove stock and provide water during the journey to eastern markets before vehicles were available for livestock transport (Ferry 1978). Elevated soil nutrient levels as a result of stock urination and defecation after heavy stocking encourages weedy annual and exotic species, which thrive in the nutrient-rich environment in periods when grazing pressure is relaxed (Lunt & Morgan 1999), hence their identification as grazing-sensitive species. The river red gum sites are unlikely to revert to their former weed-free composition now that exotic species are established at these sites, as continuing deposition of sediments and nutrients from upstream by floodwaters ensures the maintenance of elevated nutrient status. Self-reinforcing changes to nutrient cycles may also be driven by annual species (Prober et al. 2005). The community composition at these sites may therefore have moved to an alternative stable state that will require significant input in the form of weed control and buffers to reduce nutrient enrichment in runoff from surrounding cropping land in order to reduce their weed burden.

Truly grazing-sensitive species recorded in this study included woodruff (*Asperula conferta*), amulla (*Eremophila debilis*), early spring grass (*Eriochloa pseudoacrotricha*), spring grass (*E. procera*), Queensland blue grass (*Dichanthium sericeum*), curly windmill grass (*Enteropogon acicularis*), Warrego grass (*Paspalidium *jubiflorum*) and climbing saltbush. These species are highly palatable or remained green when other feed sources were unavailable, making them susceptible to livestock grazing.

In terms of biodiversity conservation value, a mosaic of land uses was important to maximise native plant conservation value across the lower Namoi catchment. Both regularly and rarely grazed sites were identified as high conservation value (Figure 6.8), with both high native plant species richness and maximum rarity. Despite this, grazing intensity was negatively related to the number of rare native plant species: increasing grazing intensity resulted in fewer rare native plants. Hence, protection of remnant vegetation from livestock grazing will be an important biodiversity conservation strategy in the study region, in future.

6.4.4 Vegetation Type and Habitat Influences on Bird Conservation Value

River red gum-dominated riparian vegetation was, on average, the most species rich vegetation type for birds, though riparian coolibah sites were also species-rich, the two vegetation types not differing significantly. Riparian sites contained uncommon species that tended not to occur across the floodplain. PCA showed that these sites were more structurally complex than other sites, with high levels of canopy cover, CWD and above-ground woody biomass. Riparian corridors of river red gum or coolibah were often forest or open-forest and were often taller than adjacent floodplain woodland. Three river red gum sites with a shrubby understorey had greater conservation value than the grassy form of the community, being ranked highly for both native species richness and maximum rarity.

River red gum sites were largely confined to the banks of the Namoi River, which provides a reliable, year-round, water supply. Drought conditions prior to and during the study and an environmental flow released from Keepit Dam on the Namoi River in September 2007 likely contributed to the species richness of riparian sites, with the bird fauna benefiting from more abundant resources. For most of the length of the river, the riparian vegetation of the Namoi is better connected than the surrounding landscape (Eco Logical 2009), acting as a corridor for wildlife and encouraging species typical of forest and humid woodland to extend westwards beyond their core distribution. The white-throated treecreeper, eastern yellow robin and white-throated warbler inhabit coastal areas and forest and woodlands of the Great Dividing Range (Higgins & Peter 2002), but occurred in the river red gum sites along the lower Namoi River. The extensive Pilliga Scrub is also close to the Namoi River corridor for a large portion of its route through the study region, providing access to extensive open-forest habitat.

Nine bird species (bar-shouldered dove, black kite, eastern yellow robin, golden whistler, little corella, rock dove, white-faced heron, white-throated treecreeper and white-throated warbler) were only recorded in river red gum sites. Of these species, the black kite, eastern yellow robin, little corella, rock dove and white-faced heron were recorded at multiple sites, while the bar-shouldered dove, golden whistler and white-throated treecreeper were only recorded once in a particularly species-rich site close to the Pilliga Scrub. The black kite, little corella and white-faced heron commonly inhabit inland waterways, while the eastern yellow robin nests in the shrub understorey commonly found along inland rivers (Higgins & Peter 2002). Rock doves are an introduced species commonly observed across NSW and other areas across Australia. It was only by chance that this species was recorded solely in river red gum vegetation: the species was likely taking advantage of surface water and nearby grain.

River red gum sites were rich with bird species due to the range of foraging opportunities provided by structurally complex vegetation and the presence of water. This community provides an abundance of nesting opportunities including silty banks for construction of burrows by pardalotes and kingfishers, dense shrub and tree foliage for small woodland birds, hollows and surface water for parrots and cockatoos, large dead branches and tall stags for raptors, and heterogeneous ground cover of CWD, litter and herbage and bare ground for wrens rufous songlarks and perch-and-pounce flycatchers. Riparian gallery forests have high fertility and productivity and hence support a much larger faunal community than adjacent floodplains and uplands (Naiman et al. 2005). Riparian vegetation communities across Australia are noted for their importance in the conservation of distinct bird communities, especially those in fragmented and semi-arid landscapes (e.g. Fischer & Goldney 1997; MacNally et al. 2000; Martin et al. 2006; Palmer & Bennett 2006; Woinarski et al. 2000), and during drought periods (Ford et al. 2001).

High bird richness in river red gum communities may be a result of the presence of both edge specialists and species from adjacent vegetation types spilling into this vegetation type (Berry 2001). Berry (2001) reported that four bird species (white-throated treecreeper, grey shrike-thrush, grey fantail and superb fairy-wren) were more abundant on forest edges and attributed this to greater abundance of foraging opportunities in the edge and adjacent, more open habitat. All four of these species were recorded in river red gum vegetation in the present study, although the superb fairy-wren was also recorded in other vegetation types, especially where scattered mimosa (*Vachellia farnesiana*), lignum (*Muehlenbeckia florulenta*), or black roly poly shrubs were present to offer protection. In addition, of the birds Howe (1984) listed as forest edge specialists, the following were recorded in the current study in river red gum vegetation: laughing kookaburra, black-faced cuckoo-shrike, white-winged triller, rufous whistler, white-throated warbler, brown thornbill, noisy friarbird, noisy miner, mistletoebird, striated pardalote, white-winged chough, dusky woodswallow and grey butcherbird. There is a lack of literature on the occurrence and impact of edge effects in woodland and open-woodland on bird community composition compared to forest ecosystems. However, these effects are likely to be less pronounced in woodlands as a result of their more open structure.

Like river red gum sites, nine bird species (brown falcon, common bronzewing, horsfield's bronze cuckoo, masked woodswallow, pacific black duck, pacific heron, rainbow bee-eater, white-breasted sea eagle and white-winged triller) were found exclusively in coolibah-dominated vegetation. Of these species only the white-breasted sea eagle and the white-winged triller were recorded at multiple sites. A pair of white-breasted sea eagles was recorded nesting high in a river red gum and hunting over adjacent coolibah sites, while white-winged trillers are commonly recorded in open semi-arid woodlands (Higgins et al. 2006) and were recorded in four sites. None of these species are coolibah woodland specialists, although they frequently occur in semi-arid inland woodlands (Higgins et al. 2006). The white-breasted sea eagle is protected under international agreements on migratory birds (CITES and CAMBA) and under Australian legislation (*EPBC Act 1999*: marine and migratory species conservation).

Of the other vegetation types, grasslands were least speciose and many of the species recorded in this vegetation type were common across the floodplain. Most of the species recorded in grasslands are successful in modified, especially agricultural, landscapes. Grassland sites were, as a result, less unique, but some species were recorded only in grassland sites, including the black falcon, brown songlark, channel-billed cuckoo, dusky woodswallow, ground cuckoo-shrike and noisy friarbird. These species were all recorded only once, the channel-billed cuckoo and noisy friarbird being recorded in isolated trees in grasslands. Many woodland vertebrate species use adjacent grasslands for hunting or foraging, making grasslands in a landscape mosaic important (Law & Dickman 1998).

6.4.5 Grazing Impacts on Site Value for Bird Conservation

The degree to which changes in vegetation composition and structure caused by livestock grazing influence native faunal assemblages remains largely unknown (Martin & Possingham 2005). However, many studies have demonstrated the effects of grazing on plant community structure (e.g. Dorrough & Moxham 2005; Fondell & Ball 2004; Harrington et al. 1984; Reid & Landsberg 2000) and the influence of structural changes in vegetation due to grazing on bird communities (Bowen & Kruse 1993; Dambach 1944; Dambach & Good 1940; Fondell & Ball 2004; Good & Dambach 1943; Jansen & Robertson 2001; Martin & Possingham 2005). In a study of the effects of livestock grazing on animal and plant biodiversity in shortgrass steppe, Milchunas et al. (1998)

found that of all the faunal groups studied (plants, lagomorphs, rodents, birds, above-ground and below-ground macroarthropods, microarthropods and nematodes), plants and birds were the most responsive to grazing.

In this study, increased grazing reduced the conservation value of sites; most rarely grazed sites were ranked highly for maximum rarity (Figure 6.15), and bird community composition was significantly impacted by grazing in river red gum sites. SIMPER analysis identified grazing-sensitive and grazing-tolerant species, three of the grazing-sensitive species being listed as declining woodland species in the sheep–wheat belt by Reid (1999). The grazing-sensitive species were predominantly small to medium-sized insectivores. Grazing-tolerant species, on the other hand, were predominantly common, omnivorous species that prefer open habitats or nest in hollows. These species (e.g. Australian ringneck, blue bonnet, sulphur-crested cockatoo, eastern rosella and cockatiel) are likely taking advantage of senescing paddock trees and dead standing trees resulting from ringbarking activities.

Jansen and Robertson (2001) found that livestock grazing degraded habitat and, as a result, impacted bird community composition in the Murray and Murrumbidgee catchments in south-eastern Australia. Reid (1999) also suggested that grazing negatively impacted bird communities in the sheep–wheat belt of south-eastern Australia. In the study by Jansen and Robertson (2001), however, differences between grazed and ungrazed sites may have been as a result of the confounding influence of vegetation type or other management variables (e.g. logging). No confounding management influence was evident in the current study.

6.4.6 Overall Conservation Value of Sites

Biodiversity conservation value was assessed using the ‘maximum rarity only’ criterion for the plant dataset and the ‘maximum rarity & species richness’ criterion for the bird dataset. Using these two criteria, seven sites were valuable for both plant and bird conservation. Three of the sites valuable for plant and bird conservation were river red gum sites, two were coolibah sites, one site was a grassland site and one was a myall site. Sites across the entire east–west extent of the study region were included, most being rarely grazed sites but one was regularly grazed. Four of the seven sites were within 2 km of the Pilliga Scrub.

When the top 16 sites were ranked to conserve the greatest number of species at a minimum of two sites, the ‘average rarity only’ criterion was the most efficient for the plant dataset and the ‘species richness only’ criterion was the most efficient for the bird data. Using these two criteria, five sites were valuable for both plant and bird conservation. Four of the five sites were rarely grazed river red gum sites along the Namoi River near Pilliga. The other site was a regularly grazed grassland site west of Narrabri. Four sites valuable for both plant and bird conservation were listed in both rankings (i.e. in the sites encompassing both one occurrence of as many species as possible and a minimum of two occurrences of as many species as possible).

6.5 Conclusion

The results of this study show that the biodiversity conservation value of remnant native vegetation in an agricultural landscape varies with site, taxon (i.e. vascular plants or birds) and management history. Various criteria can be used to assess biodiversity conservation value and the most efficient criteria vary depending on the goal and taxon. The impact of grazing was more pronounced for birds than plants, but increasing grazing intensity negatively impacted biodiversity by reducing the number of rare species and altering the composition of both plant and bird communities. This provides a conundrum for management as there are trade-offs between biodiversity conservation and livestock grazing. At most sites across the lower Namoi floodplain, bird and vascular plant conservation was incompatible with commercial livestock grazing.

Ungrazed sites provided the best opportunities for biodiversity conservation, and many of these sites are already being managed by landholders as defacto conservation or minimal use areas, their plant and bird assemblages reflecting this. Some species in river red gum and coolibah communities appear able to persist under livestock grazing as they have done so since settlement. However, other species in these vegetation types may require ungrazed areas to ensure their persistence in the landscape. The number of vascular plant and bird species only represented once in the sample of 54 sites may be a concern, and the likelihood of maintaining viable populations of these rare species under current management needs to be assessed. The impacts on biodiversity of increasing the area of native vegetation dedicated to grazing should be considered and landholders encouraged to set aside areas for conservation of both grazing-sensitive birds and plants.

Appendix 1: Spearman rank correlations between native plant species richness ranking, maximum rarity ranking, NMDS axes, the five principal components and individual environmental variables.

	Nat. rich.	Max. rare	PC1	PC2	PC3	PC4	PC5	NMDS1	NMDS2	pH	EC	Ca	Mg	K	Na	ECEC	EK1	EK2	Clay	Silt	Sand	N	Flooding
Max. rare	0.28*																						
PC1	0.29*	–0.16																					
PC2	0.08	0.42**	0.06																				
PC3	0.03	–0.15	0.02	–0.07																			
PC4	–0.08	0.05	0.07	0.08	0.07																		
PC5	–0.02	–0.43**	0.17	–0.01	–0.04	0.06																	
NMDS1	–0.15	0.27*	–0.73**	0.15	0.09	–0.29*	–0.36*																
NMDS2	–0.25	0.00	–0.10	0.44**	0.05	0.20	0.04	–0.04															
pH	0.21	–0.40**	0.72**	–0.42**	–0.02	0.18	0.24	–0.73**	–0.18														
EC	0.21	–0.25	0.81**	–0.27*	–0.01	0.16	–0.06	–0.71**	–0.22	0.73**													
Ca	–0.28*	–0.26	–0.68**	–0.51**	0.01	0.01	0.25	0.25	–0.12	–0.19	–0.45**												
Mg	–0.12	–0.05	–0.47**	–0.77**	–0.14	–0.20	–0.28*	0.28*	–0.39**	–0.08	–0.10	0.54**											
K	0.14	–0.27*	0.22	–0.51**	0.81**	–0.04	–0.06	–0.19	–0.25	0.33*	0.35*	0.06	0.23										
Na	0.08	–0.27*	0.55**	–0.56**	–0.11	0.13	–0.06	–0.55**	–0.33*	0.67**	0.79**	–0.14	0.22	0.32*									
ECEC	–0.20	–0.27*	–0.45**	–0.87**	–0.03	–0.04	–0.04	0.13	–0.34*	0.09	–0.07	0.78**	0.88**	0.30*	0.31*								
EK1	0.03	–0.17	0.41**	0.22	0.11	0.54**	0.61**	–0.59**	0.26	0.25	0.31*	–0.22	–0.46**	0.05	0.20	–0.31*							
EK2	0.20	–0.18	0.39**	–0.22	0.86**	0.01	–0.03	–0.22	–0.13	0.29*	0.38*	–0.21	–0.06	0.93**	0.24	–0.02	0.22						
Clay	–0.14	–0.28*	–0.37*	–0.89*	0.04	–0.07	–0.03	0.04	–0.30*	0.11	–0.03	0.64**	0.82**	0.36*	0.28*	0.88**	–0.28*	0.08					
Silt	–0.11	0.23	–0.39**	0.29*	0.21	0.66**	–0.25	0.25	0.40**	–0.25	–0.28*	0.12	–0.17	–0.13	–0.35**	–0.08	0.05	–0.10	–0.15				
Sand	0.23	0.08	0.57**	0.55**	–0.21	–0.41**	0.20	–0.24	0.01	0.06	0.24	–0.62**	–0.56**	–0.23	0.00	–0.68**	0.23	–0.01	–0.72**	–0.52**			
N	–0.18	0.25	–0.61**	0.42**	0.12	0.26	–0.31*	0.54**	0.32*	–0.69**	–0.52**	0.17	–0.03	–0.20	–0.49**	–0.06	–0.03	–0.13	–0.19	0.59**	–0.21		
Flooding	–0.09	0.10	–0.44**	0.09	0.01	–0.58**	0.01	0.54**	0.05	–0.43**	–0.46**	0.12	0.16	–0.17	–0.36*	0.04	–0.32*	–0.18	0.03	–0.06	0.04	0.11	
Mean rain	–0.22	0.28*	–0.73**	0.29*	–0.16	–0.27*	–0.25	0.65**	0.24	–0.72**	–0.64**	0.28*	0.17	–0.38**	–0.50**	0.07	–0.32*	–0.41**	–0.02	0.18	–0.10	0.60**	0.45**

Appendix 2: Spearman rank correlations between native bird species richness ranking, maximum rarity ranking, NMDS axes, the five principal components and individual habitat variables.

	Max. rare	PC1	PC2	PC3	PC4	PC5	NMDS1	NMDS2	Canopycov	tshrubcov	lshrubcov	herbcover	Litcover	Totground	Woodbio	Herbbio	litbio	CWDbio	Noltrees	Nostrees	Nosap	Noshrub	Nosnags	NoCWD	TreeSppRi	ShrubSppR
PC1	–0.34*																									
PC2	–0.24	–0.02																								
PC3	–0.18	–0.03	–0.09																							
PC4	–0.24	–0.01	0.05	–0.08																						
PC5	–0.04	0.02	0.14	–0.12	0.08																					
NMDS1	0.30*	–0.17	–0.54**	–0.07	0.00	–0.04																				
NMDS2	–0.39**	0.55**	–0.19	–0.02	0.23	0.15	0.05																			
Canopycov	0.24	–0.90**	0.03	0.23	0.04	0.06	0.09	–0.43**																		
tshrubcov	0.26	–0.83**	0.19	0.26	–0.27*	–0.07	–0.03	–0.59**	0.78**																	
lshrubcov	–0.17	0.10	0.27*	0.12	0.17	–0.64**	–0.19	–0.13	–0.11	0.00																
herbcover	0.03	0.42**	–0.68**	0.46**	–0.07	–0.17	0.25	0.33*	–0.35*	–0.39**	–0.12															
Litcover	0.40**	–0.58**	–0.60**	–0.13	–0.04	–0.31*	0.55**	–0.23	0.46**	0.32*	–0.08	0.09														
Totground	0.37*	–0.31*	–0.87**	0.15	–0.11	–0.29*	0.54**	–0.06	0.27*	0.13	–0.17	0.57**	0.83**													
Woodbio	0.31*	–0.81**	–0.29*	0.24	0.28*	–0.05	0.29*	–0.36*	0.81**	0.60**	–0.18	–0.04	0.61**	0.52**												
Herbbio	–0.21	0.46**	–0.33*	0.63**	–0.21	–0.21	0.04	0.23	–0.34*	–0.26	0.01	0.69**	–0.22	0.16	–0.19											
litbio	0.34*	–0.67**	–0.49**	–0.18	–0.01	–0.15	0.45**	–0.35*	0.52**	0.40**	–0.20	–0.06	0.86**	0.66**	0.65**	–0.25										
CWDbio	0.29*	–0.82*	–0.09	–0.07	0.27*	–0.08	0.24	–0.43**	0.70**	0.59**	–0.12	–0.28*	0.49**	0.31*	0.79**	–0.38*	0.57**									
Noltrees	0.14	–0.58**	–0.39**	0.31*	0.43**	–0.05	0.29*	–0.06	0.62**	0.40**	–0.11	0.06	0.51**	0.48**	0.83**	–0.04	0.51**	0.61**								
Nostrees	0.21	–0.87**	0.27*	0.02	–0.14	0.01	–0.07	–0.57**	0.80**	0.78**	0.00	–0.51**	0.34**	0.05	0.58**	–0.41**	0.45**	0.65**	0.32**							
Nosap	0.28*	–0.79**	0.15	0.17	–0.34*	–0.01	0.08	–0.53**	0.75**	0.83**	–0.04	–0.39**	0.35*	0.12	0.55**	–0.28*	0.45**	0.51**	0.30*	0.79**						
Noshrub	0.00	–0.45**	0.46**	0.38*	–0.24	–0.35**	–0.20	–0.48**	0.42**	0.64**	0.26	–0.34*	0.05	–0.13	0.23	–0.03	0.05	0.30*	0.11	0.49**	0.48**					
Nosnags	0.21	–0.79**	0.12	–0.24	0.05	–0.02	–0.01	–0.48**	0.63**	0.60**	–0.10	–0.41**	0.36*	0.13	0.57**	–0.45**	0.51**	0.74**	0.35*	0.77**	0.58**	0.26				
NoCWD	0.14	–0.80**	–0.05	–0.01	0.28*	–0.04	0.15	–0.28*	0.72**	0.58**	–0.14	–0.33*	0.39**	0.22	0.75**	–0.30*	0.53**	0.83**	0.61**	0.65**	0.49**	0.28*	0.67**			
TreeSppRi	–0.11	–0.47**	0.14	0.30*	0.32*	0.32*	0.02	–0.20	0.50**	0.39**	0.04	–0.20	0.12	0.01	0.41**	–0.26	0.25	0.30*	0.39**	0.36*	0.39**	0.22	0.28*	0.29*		
ShrubSppR	0.03	–0.47**	0.50**	0.02	0.23	–0.08	–0.26	–0.39**	0.37*	0.41**	0.12	–0.39**	–0.05	–0.22	0.28*	–0.32*	0.05	0.34*	0.10	0.42**	0.30*	0.48**	0.51**	0.33*	0.30*	
natrichra	0.68**	–0.69**	–0.15	0.06	–0.11	–0.11	0.25	–0.59**	0.62**	0.59**	–0.12	–0.14	0.49**	0.39**	0.68**	–0.30*	0.52**	0.59**	0.43**	0.51**	0.57**	0.33*	0.45**	0.47**	0.18	0.27*

Chapter 7: Comparison of Ecosystem Services provided by Different Vegetation Types and Condition States: Trade-offs with Management and Land Use Intensity

Cattle grazing native pasture in a coolibah (*Eucalyptus coolabah*) woodland near Pilliga (May 2008).

Chapter 7: Comparison of Ecosystem Services Provided by Different Vegetation Types and Condition States: Trade-offs with Management and Land Use Intensity

7.1 Introduction

The literature on ecosystem services has grown exponentially over the last two decades (De Groot et al. 2002). The term now appears in policy documents of the United Nations, governments and non-government organisations, and protection of ecosystem services has been used as justification for investment in conservation and land use change (e.g. the protection of the Catskill Mountains catchment to maintain the quality of New York's water supply: Heal 2000). Ecosystem services are also the focus of many global-scale research projects (e.g. Costanza et al. 1997; Naidoo et al. 2008) and international collaborative efforts (e.g. MA 2005). Unfortunately, the ecosystem services concept has spread rapidly and the research required to inform policy and management decisions at a landscape or site scale has been unable to keep pace, with the exception of knowledge of many provisioning services (e.g. Balmford et al. 2002; MA 2005; Naidoo et al. 2008). Despite the large body of literature referring to the benefits that nature provides to people, but there is insufficient quantitative data on ecosystem service provision in most ecosystems (Balvanera et al. 2001; Carpenter et al. 2009; Daily 1997; Kremen 2005; Nelson et al. 2009). The effects of land use and variations in ecosystem composition, structure and function on ecosystem service provision are largely unquantified (Bennett et al. 2009; DeFries et al. 2004; Foley et al. 2005; Kremen 2005; Naidoo et al. 2008; Nelson et al. 2009), and there is even less information on the potentially complex spatial and temporal interrelations between multiple ecosystem services (DeFries et al. 2004; Heal et al. 2001; Rodríguez et al. 2006). Quantitative data is urgently required to inform land use and management decisions, minimise trade-offs and maximise synergies in multiple ecosystem service provision (Bennett et al. 2009; Carpenter et al. 2006; DeFries et al. 2004; Farber et al. 2006; Kareiva et al. 2007; Nelson et al. 2009; Rodríguez et al. 2006).

Currently, research that quantifies ecosystem service provision falls into two groups, the ‘benefit transfer’ approach and the ‘ecological production function’ approach (Nelson et al. 2009). The benefit transfer approach includes studies such as Costanza et al. (1997) that use limited estimates of ecosystem service provision in various biomes to extrapolate values across large areas or the globe. Naidoo et al. (2008) also followed the benefit transfer approach using coarse, global-scale data to determine high priority sites for simultaneous biodiversity and ecosystem service conservation. These approaches assume every hectare of each ecosystem provides the same level of ecosystem service provision. At the other end of the spectrum, the ecological production function approach quantifies a single service under a narrow range of ecological conditions. The latter approach yields detailed and reliable data but is limited in its application to one service (and is therefore not comprehensive) and to a specified range of conditions (and therefore lacks general applicability) (Nelson et al. 2009). Few studies quantify ecosystem service production functions that describe the relationships between ecosystem condition and service provision (Carpenter et al. 2006; Nelson et al. 2009). Biophysical data on ecosystem service provision is required at a scale that is useful to managers.

7.1.1 Biodiversity and Ecosystem Service Provision

The relationship between ecosystem functioning and species richness or diversity is a continuing source of debate (e.g. Swift et al. 2004). A great deal of research has been directed at determining the relationship between the two, with varying models proposed to explain the relationship (e.g. the ‘idiosyncratic’ model: Lawton 1994; the ‘rivet’ hypothesis: Ehrlich & Ehrlich 1981, and the redundant species model: Walker 1992). Most studies have surmised a reduction in ecosystem function with biodiversity loss. Up to 95% of the experiments reviewed by Schwartz et al. (2000) supported the redundant species hypothesis, which postulates that a threshold effect exists between ecosystem function and biodiversity: once a threshold in biodiversity loss is reached, ecosystem functioning ceases or is greatly impaired. However, the relevance of such studies is questionable as they are restricted to a limited set of ecosystem functions (Schwartz et al. 2000) and the structure and composition of experimental communities differ markedly from real landscapes (e.g. Symstad et al. 2003).

By comparison, little effort has been directed towards determining the relationship between species richness or diversity and ecosystem service provision (Tscharntke et al. 2005). The level of congruence between the provision of multiple ecosystem services and biodiversity is yet to be tested (Chan et al. 2006; Naidoo et al. 2008; Nelson et al. 2009). The ecosystem services concept is purported to be useful in achieving broad-scale biodiversity conservation benefits, yet there is little evidence that this is the case (McCauley 2006; Redford & Adams 2009) Quantitative data are rare and support opposing conclusions, with results ranging from little to no association between biodiversity and ecosystem service provision (e.g. Chan et al. 2006; Egoh et al. 2007;

Naidoo et al. 2008) to strong relationships (e.g. Nelson et al. 2009). Information is required to operationalise the principle that human welfare and prosperity are linked to biodiversity, and that sustainable development must consider and manage for both biodiversity and ecosystem service provision (Naidoo et al. 2008).

7.1.2 The 'Ecosystem Service Providers' Concept

The functional mechanisms behind ecosystem service provision are complex, arising from interdependent components and non-linear interactions (Carpenter et al. 2006), hence very little is known of the ecology of ecosystem service provision (Kremen 2005). Kremen (2005) proposed a research agenda for the management of ecosystem services based on identification and management of ecosystem service providers (ESPs). Ecosystem service providers are the component populations, species, functional groups (guilds), food webs or habitat types that collectively generate ecosystem services (Kremen 2005). Kremen's research agenda consisted of five steps: (1) identifying the species or other entities that are key ESPs and characterising their functional relationships; (2) determining the various aspects of community structure that influence function in real landscapes, especially compensatory community responses that stabilise function, or non-random extinction sequences that rapidly erode it; (3) assessing the key environmental factors influencing provision of services, and (4) measuring the spatio-temporal scale over which providers and services operate.

The 'functional importance' of an ESP is dependent on its efficacy in providing the service and its abundance in the ecosystem; entities that are disproportionately important relative to their abundance are said to have high functional importance

(Balvanera et al 2005). These entities become the target for management, as their persistence is paramount to the provision of the ecosystem service. Another useful measure is the ‘functional attribute diversity’ of a community or the ‘ecological distance’ between ESPs in functional attribute space (Walker et al. 1999). This method highlights functionally redundant species and thus the resilience of the ecosystem to disturbance.

A related concept, the service providing unit (SPU) refers to the individuals from a given species and their characteristics necessary to deliver an ecosystem service at a given level (Luck et al. 2003). The SPU and ESP concepts were combined in 2009 and referred to as the ‘service provider’ continuum (Luck et al. 2009). The SPU–ESP continuum promoted the quantification of organism, community or habitat characteristics required to provide ecosystem services in light of beneficiary demands and ecosystem dynamics (Luck et al. 2009). The ESP concept focuses on the broader level of vegetation communities and plant functional groups as ecosystem service providers, whereas the SPU concept is confined predominantly to segments of a population or populations that provide ecosystem services.

7.1.3 Vegetation Condition Assessment for Ecosystem Service Provision

Vegetation condition is a context-dependent concept and often related to societal values (Oliver et al. 2002). For example, a grazier may assess the condition of a paddock in terms of its productive and nutritive value for livestock, its value for shade and shelter, and more recently, perhaps, carbon sequestration value. A forester may assess vegetation condition in terms of wood production, and a park ranger may assess departure from some reference state in terms of ecosystem composition, structure and function (biodiversity). Rarely is vegetation condition assessed in terms of ecosystem service provision, except where there is a direct economic value involved (i.e. provisioning services). However, biodiversity conservation value assessments are carried out occasionally on the premise that more biodiversity will result in greater ecosystem service provision (see Egoh et al. 2009 for a review).

7.1.4 Grazing Effects on Ecosystem Service Provision

Grazing is the most widespread land use in the world (FAO 2002) and has the potential to alter ecosystem composition, structure and function to the extent that it rivals the effects of cropping in intensive situations (Benson 1991; Wilson 1990). Grazing effects on plant and bird biodiversity conservation were examined in detail in Chapter 6. Grazing had a significant effect on biodiversity conservation value for both plants and birds by altering vegetation composition and structure. Flow-on effects from grazing as a result of transformed vegetation may result in altered ecosystem functioning (Milchunas & Lauenroth 1993) and trade-offs in provision of other ecosystem services, especially where the abundance or efficiency of ESPs are affected (Kremen 2005).

Grazing affects C storage through changes in plant productivity and biomass partitioning between above and below-ground structures (Milchunas & Lauenroth 1993). Grazing modifies ecosystem composition, structure and function, and changes soil physico-chemical properties and decomposition processes (Graetz & Tongway 1986; Greene & Tongway 1989; Greenwood & McKenzie 2001; King & Hutchinson 1976; Milchunas & Lauenroth 1993). Indirect management associated with grazing such as ring-barking and clearing can be detrimental to carbon storage value by removing woody biomass: In other situations, grazing can influence woody vegetation regeneration in a positive (Harrington et al. 1979; Pressland 1984) or negative (Dorrough & Moxham 2005; Reid & Landsberg 2000) manner. Livestock grazing alters food availability, hydrology, soil temperature regimes and pore size distribution, thus affecting soil biota (King & Hutchinson 1976). Changes in soil biota result in altered decomposition rates, thus influencing soil organic matter dynamics and carbon storage (Lavelle & Spain 2001)

Grazing markedly increases the potential for soil erosion in most environments. Many studies have illustrated the effects of grazing on soil-related functions and hence erosion. Some effects include (but are not limited to) loss of litter cover, cryptograms, nutrients, soil microtopography, reduced organic carbon and soil water infiltration, soil compaction, reduced aggregate stability, and subsequent loss of ecosystem function and soil loss (e.g. Beadle 1948; Ludwig et al. 1997; Eldridge & Koen 1993; Graetz & Tongway 1986; Greene & Tongway 1989; Greenwood & McKenzie 2001; Ludwig et al. 1994; Milchunas & Laurenroth 1993; Tongway & Ludwig 1994). However, the impact of grazing on soil erosion is not always straightforward. One study by Taddesse et al. (2002) indicated that the greatest influence of grazing by livestock on vertosols in Ethiopia was in relation to soil compaction on wet or waterlogged soils and the resultant impediment to infiltration. This resulted in a reduction in the shrink–swell nature of the soil and increased volume of runoff and hence the risk of erosion. However, reduced vegetative biomass under heavy grazing resulted in enhanced soil cracking, which restarted the shrink–swell cycle and reduce the erosion hazard.

7.1.5 Ecosystem Service Provision on the Inland Floodplains of the Namoi River

Measurements of biodiversity conservation, carbon storage and erosion mitigation value of sites representing five native vegetation types common to the lower Namoi floodplain were reported in previous chapters. Briefly, bird conservation, carbon sequestration and erosion mitigation value of sites was related to the abundance of woody vegetation present. Bird conservation value was highest at sites with high canopy cover and structural complexity in understorey layers. Carbon storage was highest in sites with maximum woody biomass. Erosion mitigation, in terms of aggregate stability, was highest in sites with high soil C:N ratios, which resulted from the deposition and decomposition of lignified plant components, namely sclerophyllous plant (eucalypt) litter and coarse woody debris (CWD). Moisture availability was a major determinant of woody vegetation presence and abundance, but management also played a part in terms of woody vegetation removal and alteration of community composition and structure. Increasing grazing intensity reduced site value for plant and bird conservation value resulting in a tradeoff between nature conservation and livestock production. Trade-offs between ecosystem services and land use are little understood in highly modified agricultural environments, hence assessment of the magnitude of service provision with increasing land use intensity is required.

7.1.6 Aims and Objectives

Trade-offs among multiple ecosystem services are inevitable because not all vegetation types can provide high levels of all ecosystem services. For example, grasslands cannot produce wood, and closed-forests are poor producers of herbaceous forage. Trade-offs in ecosystem service provision with alternate land uses are more significant, especially where the vegetation is completely removed or substantially altered. Few data compare provision of multiple ecosystem services from the one vegetation or ecosystem type across varying intensities of land use. The relative importance of species richness and functional richness to ecosystem service provision is similarly uncertain, as are guidelines for assessment of vegetation condition in the context of ecosystem service provision. Accordingly, the objectives of this study were to answer the following questions:

1. What congruence exists between vegetation species richness or diversity and the provision of carbon storage, erosion mitigation and bird conservation services?
2. Does vegetation functional richness influence carbon storage, erosion mitigation or bird conservation value?
3. What are the ecosystem service providers (ESPs) for erosion mitigation, carbon storage and bird conservation value?
4. How should condition be assessed for ecosystem service provision?
5. How does grazing intensity influence carbon storage, erosion mitigation, and bird and vascular plant conservation value?
6. What are the trade-offs between grazing and carbon storage, erosion mitigation, and bird and plant conservation services?

7.2 Methods

7.2.1 Study Region

The lower Namoi floodplain in northern NSW, Australia (Figure 7.1), is a highly modified agricultural area used for livestock grazing, wheat and irrigated cotton production. Large tracts of native and derived grasslands and open woodlands are grazed extensively by sheep and cattle. Soils are predominantly grey cracking clays (vertosols), although sandy ridges (prior stream formations) feature sporadically amongst the grey clay matrix. The brown clays and sandy ridges often have sodic surface soils and always have sodic subsoils making them liable to scalding. Rainfall across the region varies from 600 mm in the east to 400 mm in the west, with a slight summer dominance. Long-term average rainfall is supplemented by summer floods that inundate the floodplain at irregular intervals. Due to the low gradient of the floodplain (1:2000), slow-moving floodwaters remain in the area for several weeks.

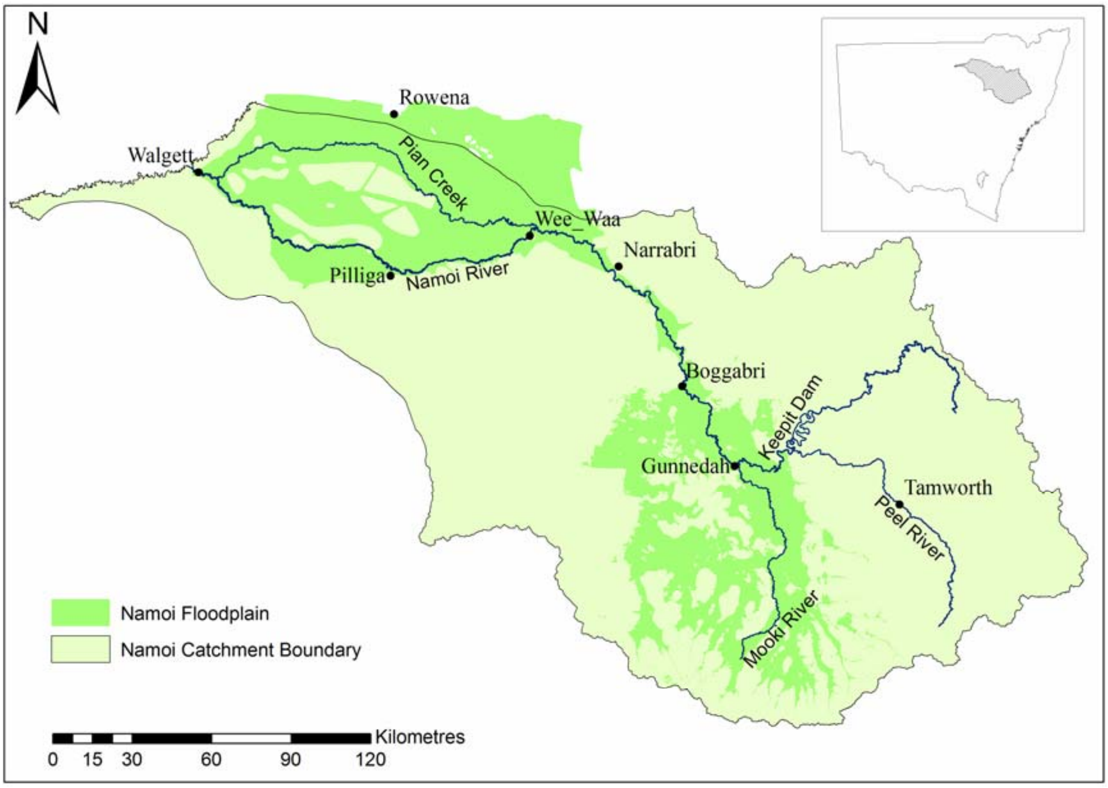


Figure 7.1: The Namoi catchment in northern NSW showing the extent of the Namoi River floodplain. The floodplain between Boggabri in the east and Walgett in the west forms the study region for this study.

Approximately 7% of the study region is covered by woody vegetation, predominantly open-woodland or scattered trees (<10% projective foliage cover). The riverine corridor is dominated by river red gum (*Eucalyptus camaldulensis*) woodland and open-forest (10–50% projective foliage cover). The floodplain open-woodlands are dominated by coolibah (*E. coolabah*) interspersed among native grasslands on the heaviest textured soils, while myall (*Acacia pendula*) tall shrublands occur on sandy ridges and black box (*E. largiflorens*) occurs on less frequently flooded brown clay soils. Derived grasslands occur on all soil types, although their floristic composition is distinct from native grasslands, especially at recently cleared sites.

7.2.2 Ecosystem Service Data

Ecosystem service provision data was collected at sites covering the range of land uses (rarely, intermittently and regularly grazed) encountered in each vegetation type, and as many structural and compositional variants as possible, from rarely disturbed, old-growth, conservation areas, to regrowth woodlands and grasslands derived though ringbarking and woody vegetation clearing. Lack of representation of some grazing intensities for some vegetation types resulted in an unbalanced survey design. Site rankings for ecosystem service provision value took the following forms.

Biodiversity conservation value was assigned by ranking sites by native plant and bird species richness and maximum rarity (Chapter 6). Birds and plants were considered separately as bird conservation value was determined by vegetation and management variables, whereas vascular plant conservation value was determined predominantly by environmental (soil and rainfall) and management variables. Carbon storage value was determined after measuring the C stored at sites in woody and herbaceous vegetation, litter, standing dead wood, coarse woody debris (CWD) and soil (Chapter 4). Sites were ranked here according to the total amount of stored C. Erosion mitigation value was based on measurement of soil aggregate stability (which was determined by soil C:N ratio, higher C:N ratio was equated with higher aggregate stability) and ground cover (herbaceous understorey and litter cover) (Chapter 5). Sites were ranked here according to values for both variables.

7.2.3 Grazing Intensity

Landholders were interviewed for information on historical and current grazing management at each site. Quantitative data were unavailable at many sites, especially on travelling stock routes (TSRs). Sites that had not been grazed by domestic livestock in the last 10 years were classified as having low grazing pressure. Sites that had been grazed in the last 10 years, but not in the last 5 were classified as having intermittent grazing pressure. Intermittently grazed sites were opportunistically crash grazed, especially in good seasons to take advantages of pulses in biomass production and therefore experienced high densities of stock for short periods of time through the year. Regularly grazed sites experienced grazing by domestic stock year round. Native macropod grazing was not considered in this study.

7.2.4 Statistical analysis

Parametric AOV of ranks was used to highlight differences in ecosystem service provision between different vegetation types, and distinguish total vascular plant species richness and functional richness (the number of functional groups or life-forms: large trees, small trees, saplings, tall shrubs, low shrubs, perennial grasses, perennial forbs, annual grasses, annual forbs and graminoids) between vegetation types. Total species richness was used in this context (i.e. both native and exotic species) as a plant doesn't necessarily need to be native to perform a function or service. Spearman rank correlations highlighted relationships between site ranking for ecosystem service provision, species richness, functional diversity and a range of potential ESPs. The ESPs investigated in this study were canopy (i.e. tree), tall shrub, herbaceous, litter and CWD cover, and woody, herbaceous, litter, CWD and dead standing biomass. Principal components analysis (PCA) was used to ordinate sites in relation to abundance of potential ESPs and bubble plots showed patterns in ecosystem service provision rankings between sites and vegetation types. The effects of grazing on ecosystem service provision were tested for river red gum and coolibah communities using AOV. These vegetation types were used as a reasonably balanced design was achieved in terms of grazing intensity, i.e. all three grazing intensities were represented among sites in these two vegetation types.

7.3 Results

River red gum sites had greater total species richness of vascular plants than coolibah and grassland sites, while myall and black box had intermediate total species richness ($F_{4,49} = 3.19$, $P = 0.02$) (Table 7.1). Functional richness was highest in river red gum and coolibah sites compared to grasslands, while myall and black box sites had intermediate functional richness ($F_{4,49} = 10.90$, $P < 0.001$). Shannon–Wiener diversity was not significantly different between vegetation types ($F_{4,49} = 2.21$, $P = 0.08$). Plant conservation value was higher in river red gum sites compared to coolibah, all other sites having intermediate plant biodiversity conservation value ($F_{4,49} = 2.85$, $P = 0.03$). While AOV suggested a significant difference between vegetation types in bird conservation value ($F_{4,49} = 2.62$, $P = 0.05$), Tukey's multiple comparisons failed to separate vegetation types on this response variable. Carbon sequestration value was significantly higher in river red gum sites than all other vegetation types and lowest in grassland sites ($F_{4,49} = 28.7$, $P < 0.001$). River red gum sites also had higher erosion mitigation value than other vegetation types ($F_{4,49} = 6.83$, $P < 0.001$).

Table 7.1: Mean total plant species richness, functional diversity and ecosystem service provision rankings by vegetation type.

	River red gum	Coolibah	Grassland	Myall	Black box
Total plant species richness	39.08a	21.35b	22.80b	22.83ab	29.25ab
Plant functional richness	31.33a	30.47a	8.47b	18.50ab	19.25ab
Shannon–Wiener diversity	28.00a	26.35a	21.40a	31.33a	45.75a
Plant conservation value	38.17a	22.00b	29.40ab	24.33ab	16.50ab
Bird conservation value	33.67a	30.77a	18.87a	27.00a	17.75a
Carbon storage	45.42a	33.35b	10.73d	24.33bc	16.50cd
Erosion mitigation	43.17a	28.71b	20.80b	16.33b	19.00b

[#] All variables are inverse ranks: 1 = lowest provision, 54 = highest provision of service. Letters represent significantly different groups as separated by Tukey's HSD ($P < 0.05$).

Site rankings for ecosystem service provision were correlated for most pairs of services (Table 7.2). Erosion mitigation value was re-calculated after excluding river red gum sites in lieu of their consistently higher erosion mitigation value and total species richness. Plant and bird conservation value, and bird conservation value and the re-calculated erosion mitigation value were not correlated. Total plant species richness was correlated with Shannon–Wiener diversity, plant conservation value, C storage and erosion mitigation, but not with the re-calculated erosion mitigation value. Plant functional richness was correlated with bird conservation value, C storage and erosion mitigation value. The correlation between erosion mitigation value and plant functional richness was non-significant for the re-calculated erosion mitigation rankings. The Shannon–Wiener diversity index was not correlated with provision of any service.

Table 7.2: Spearman correlation matrix for ecosystem services, total species richness and functional diversity.

	Plant species richness	Functional richness	Shannon-Wiener	PLANTCON	BIRDCON	CARBON
Functional richness	0.18					
Shannon–Wiener	0.46**	0.12				
PLANTCON[1]	0.48**	0.14	0.10			
BIRDCON[2]	0.01	0.37*	–0.07	0.15		
CARBON[3]	0.31*	0.64**	0.04	0.29*	0.57**	
EROSION[4]	0.38*	0.39**	0.16	0.49**	0.36*	0.67**
EROSION[#]	0.24	0.24	0.13	0.38*	0.30	0.56*

[1]site ranking for vascular plant conservation; [2]site ranking for bird conservation; [3]site ranking for carbon storage; [4]site ranking for erosion mitigation; [#]River red gum data was removed prior to computation of variable; *P ≤ 0.05; **P ≤ 0.01.

Spearman's rank correlations highlighted relationships with biomass or cover of potential ESPs for erosion mitigation value, C storage value and bird conservation value (Tables 7.3, 7.4). Woody biomass and canopy cover were positively correlated with the biomass and cover of litter, and biomass of CWD and dead standing vegetation. Herbaceous biomass was negatively correlated with woody biomass and not significantly correlated with any of the three ecosystem services. Cover of all vegetation attributes was positively correlated with site rankings for the three ecosystem services and with other vegetation attributes. However, tall shrub cover was not correlated with herbaceous vegetation or litter cover, nor erosion mitigation value.

Table 7.3: Spearman correlation matrix for ecosystem services and biomass of potential ESPs.

	Woody biomass	Herbaceous biomass	Litter biomass	CWD biomass	Dead standing biomass
Herbaceous biomass	–0.17				
Litter biomass	0.65**	–0.26*			
CWD biomass	0.79**	–0.33*	0.57**		
Dead standing biomass	0.64**	–0.33*	0.46**	0.67**	
BIRDBIO	0.55**	–0.23	0.42**	0.50**	0.40**
CARBON	0.91**	–0.14	0.71**	0.71**	0.57**
EROSION	0.53**	–0.05	0.58**	0.35*	0.39**

*P ≤ 0.05; **P ≤ 0.01.

Table 7.4: Spearman correlation matrix for ecosystem services and cover of potential ESPs.

	Canopy cover	Tall shrub cover	Herbaceous cover	Litter cover	CWD cover
Tall shrub cover	0.54**				
Herbaceous cover	0.29*	0.03			
Litter cover	0.48**	0.21	0.83**		
CWD cover	0.79**	0.46**	0.27*	0.47**	
BIRDBIO	0.50**	0.37*	0.41**	0.47**	0.44**
CARBON	0.67**	0.29*	0.69**	0.74**	0.67**
EROSION	0.38*	0.17	0.73**	0.62**	0.39**

$*P \leq 0.05$; $**P \leq 0.01$.

A PCA of the vegetation attribute data reflected the Spearman correlations (Figure 7.2). Canopy related variables (canopy cover and woody biomass, litter biomass and cover, CWD biomass and cover) were correlated with the first PC (PC1), while attributes related to herbaceous vegetation featured on the second PC (PC2) and tall shrub cover featured highest on the third PC (PC3) (Table 7.5). The first PC axis accounted for 49% of the variation in the dataset, the second PC accounting for a further 13.8% and the third an additional 10.7%. Site rankings for bird biodiversity conservation value, C sequestration value and erosion mitigation value were overlaid on the PCA to illustrate site value for each of the individual ecosystem services. Sites decreased in value for each of the three ecosystem services from left to right across the PCA ordination when PC1 and PC2 were plotted. The placement of sites along the first PC was similar to that reported in Chapter 6 and represented a moisture availability gradient. Sites on the left side of the PCA had greater available water than sites on the right.

Table 7.5: Five principal components representing linear combinations of 18 habitat characteristics and the variance explained by each.

	PC1	PC2	PC3	PC4	PC5
Eigenvalues	4.90	1.38	1.07	0.98	0.60
Variance (%)	49.00	13.80	10.70	9.80	6.00
Cumulative Variation (%)	49.00	62.80	73.50	83.20	89.30

	Code	Habitat Characteristic Factor Loadings				
Canopy cover	CC	–0.33	–0.24	0.26	0.24	–0.04
Tall shrub cover	TSC	0.08	–0.41	–0.71	–0.27	0.41
Herbaceous cover	HC	–0.33	0.45	–0.24	–0.23	–0.08
Litter cover	LC	–0.38	0.21	–0.07	–0.43	–0.23
Coarse woody debris cover	CWC	–0.34	–0.17	0.36	0.04	0.54
Woody biomass	WB	–0.36	0.10	–0.25	0.22	0.22
Herbaceous biomass	HB	0.14	0.64	–0.14	0.42	0.42
Litter biomass	LB	–0.42	0.10	0.03	–0.24	0.03
Coarse woody debris biomass	CWB	–0.37	–0.19	–0.03	0.19	0.18
Dead standing biomass	DSB	–0.25	–0.16	–0.39	0.56	–0.48

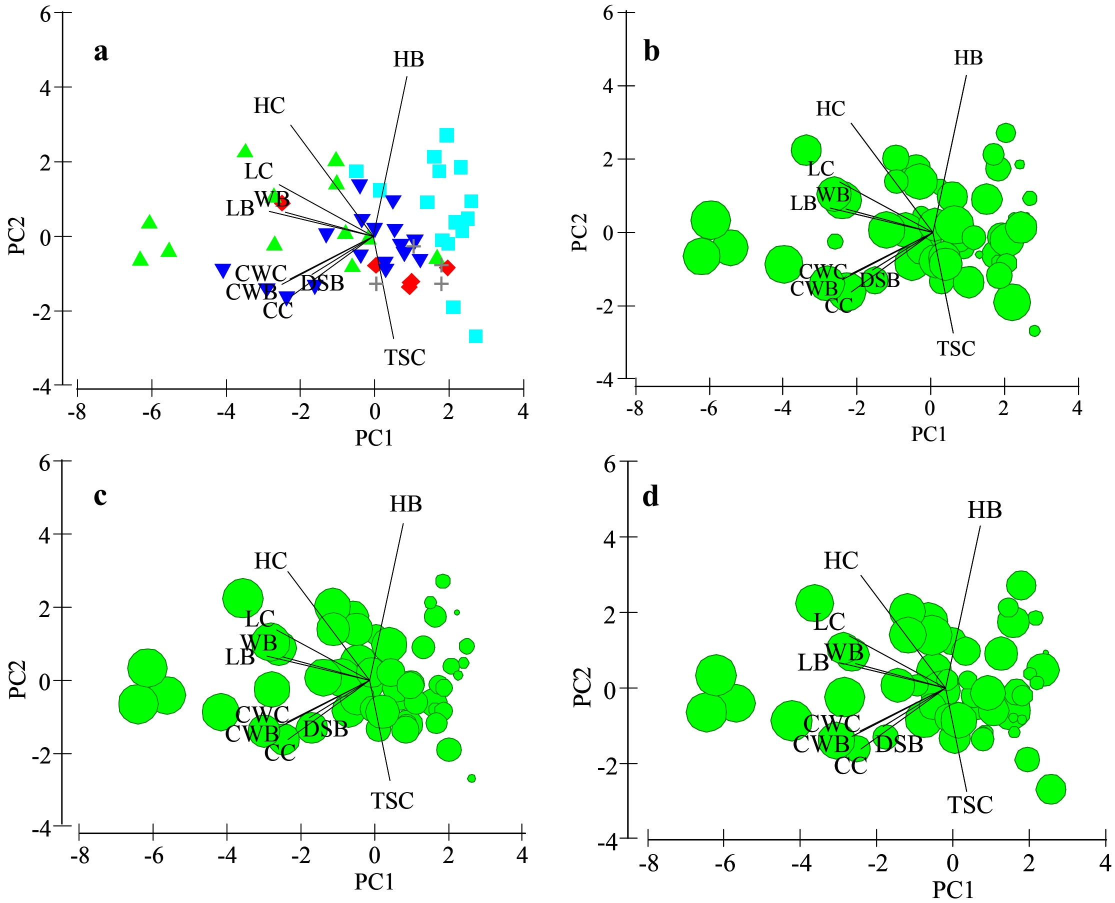


Figure 7.2: (a) PCA of ESPs with (b) bird conservation value, (c) carbon storage value and (d) erosion mitigation value represented by increasing bubble size. Vector codes shown in Table 7.5. (▲ = river red gum, ▼ = coolibah, ♦ = myall, + = black box, ■ = grassland).

Significant relationships were detected between grazing intensity and canopy cover ($F_{2,26} = 4.68$, $P = 0.018$), tall shrub cover ($F_{2,26} = 4.00$, $P = 0.031$) herbaceous cover ($F_{2,26} = 5.52$, $P = 0.010$), litter cover ($F_{2,26} = 8.23$, $P = 0.002$) and litter biomass ($F_{2,26} = 7.44$, $P = 0.003$) in river red gum and coolibah communities (Table 7.6). These differences had flow-on consequences for plant conservation value ($F_{2,26} = 3.76$, $P = 0.037$) and bird conservation value ($F_{2,26} = 5.17$, $P = 0.013$) and carbon storage ($F_{2,26} = 3.34$, $p = 0.051$), but not erosion mitigation ($F_{2,26} = 1.71$, $P = 0.200$). While not always significant, all vegetation attributes in river red gum and coolibah except herbaceous biomass (which showed no relationship) were negatively influenced by increased grazing pressure, as were the rankings of the provision of all ecosystem services (Table 7.6).

Table 7.6: Grazing impact on ESPs and ecosystem service provision in river red gum and coolibah vegetation types.

	Grazing intensity Regular	 Intermediate	 Low
Total plant species richness[#]	12.31a	17.86a	15.22a
Plant functional richness[#]	9.31a	10.14a	14.00a
Canopy cover (%)	17.62b	19.86ab	33.33a
Tall shrub cover (%)	0.98b	1.17ab	2.24a
Herbaceous cover (%)	56.04b	72.39ab	78.35a
Litter cover (%)	25.20b	43.67ab	52.10a
CWD cover (%)	2.51a	2.86a	3.90a
Woody biomass (t ha^{-1})	86.21a	128.13a	201.10a
Herbaceous biomass (t ha^{-1})	1.48a	1.39a	1.52a
Litter biomass (t ha^{-1})	1.82b	2.95ab	3.44a
CWD biomass (t ha^{-1})	4.14a	5.15a	10.53a
Dead standing biomass (t ha^{-1})	2.89a	5.23a	9.21a
Plant conservation value[#]	11.31b	14.71ab	20.56a
Bird conservation value[#]	11.54b	12.71ab	21.56a
Carbon storage[#]	11.39b	15.00ab	20.22a
Erosion mitigation[#]	11.85a	15.71a	18.89a

[#] Variables are site inverse rankings for ecosystem service provision: 1 = lowest provision, 54 = highest provision of service. Letters denote significantly different groups as defined by Tukey's HSD ($P \leq 0.05$).

7.4 Discussion

7.4.1 Relationship between Biodiversity and Ecosystem Service Provision

Several authors have expressed the need to investigate the level of congruence between biodiversity conservation and ecosystem service provision (e.g. Chan et al. 2006; MA 2005; Naidoo et al. 2008; Nelson et al. 2009). However, the few studies to date have come to opposing conclusions, depending on the ecosystem services under investigation (e.g. Egoh et al. 2009; Naidoo et al. 2008). Conservation planning agencies are also beginning to include ecosystem service considerations in reserve design and prioritisation, but potential trade-offs or synergies between biodiversity and ecosystem services are not fully known, compromising resource allocation (e.g. Egoh et al. 2007, 2009).

In the current study, plant and bird conservation value were significantly correlated with C storage and erosion mitigation services. However, plant conservation value and bird conservation were not significantly correlated with each other, and when river red gum sites were removed, bird conservation value was not correlated with erosion mitigation. This suggests that there is a high degree of congruence between plant conservation value and C storage and erosion mitigation, but that the nature of the synergy is complex.

Across all sites and vegetation types, total plant richness was correlated with native plant conservation value, C storage and erosion mitigation, but not bird conservation value. However, within individual vegetation types, plant conservation value only was correlated with total plant species richness in coolibah ($r_s = 0.549$, $P = 0.024$) and black box ($r_s = 1.000$, $P < 0.001$) communities. No other ecosystem service in any vegetation type was correlated with total plant richness. Plant species conservation value was determined by the maximum rarity index (i.e. sites were ranked highest for plant conservation value where the number of rare native species was highest). The lack of a significant relationship between total plant species richness and plant conservation value in other vegetation types indicated that total plant richness in these communities was driven by exotic species richness and common native species (see also Chapter 6), not rare plant species.

The overall relationship between total plant species richness and C storage was driven by river red gum sites. River red gum sites were consistently ranked among the highest for both total species richness and C storage value. Riparian areas worldwide are renowned for being species rich and productive ecosystems as a result of elevated fertility and moisture availability (Naiman et al. 1993, 2005). When river red gum sites were removed and correlations re-calculated on the remaining vegetation types, there was no significant relationship between total plant species richness rank and C storage value ($r_s = 0.022$, $P = 0.891$). No significant correlation was found between total species richness rank and C storage in any of the individual vegetation types. The lack of a relationship between species richness and C storage value contradicts with experimental evidence of a significant relationship between species richness and diversity and ecosystem productivity (e.g. Tilman et al. 1996; Hector et al. 1999). However, the current study was not conducted in small, uniform plots as in the experiments to date, but on complex assemblages of species in natural or semi-natural environments spanning several vegetation types. The added considerations of grazing management (discussed later on), and differences in soil physico-chemical characteristics and moisture availability need to be borne in mind in the current study.

When the erosion mitigation value was recalculated without river red gum sites, the relationship between total plant species richness and erosion mitigation was no longer significant. No relationship between total species richness and erosion mitigation value was found in any of the individual vegetation types. The erosion mitigation value of sites was measured using ground cover (the sum of herbaceous and litter cover) and aggregate stability (C:N ratio). River red gum sites have high aggregate stability as a result of large inputs of sclerophyllous leaf litter and CWD over long timeframes (i.e. from centuries-old river red gum trees) and elevated total species richness. These organic matter inputs to the soil have a high C:N ratio as a result of the high lignin content (Lavelle & Spain 2001) and stabilise soil macroaggregates.

7.4.2 Influence of Vegetation Functional Richness on Ecosystem Service Provision

Vegetation functional richness showed strong relationships with ecosystem service provision. Carbon storage was particularly influenced by functional richness, though all ecosystem services showed stronger relationships with functional richness compared to species richness. Grasslands were the least functionally rich while river red gum and coolibah sites were the most functionally rich.

Vegetation structure is important for bird communities in providing nesting, perching and foraging opportunities, and protection from predators (e.g. Bowen & Kruse 1993; Fondell & Ball 2004; Jansen & Robertson 2000; MacArthur & MacArthur 1961; Martin

& Possingham 2005), hence the relationship between vegetation functional richness and bird conservation value. Complex vegetation structure is provided by functional groups such as large hollow-bearing trees, dense shrub mid-storey and ground cover. However, there were sites such as some grasslands that were highlighted in the PCA ordination as important for bird conservation, since bird conservation value was measured using rarity of individual species as well as species richness. Some bird species specific to more open habitat (i.e. with less functional richness), for example the ground cuckoo-shrike and Australian pipit were only recorded in grasslands. This emphasises the need for a mosaic of habitat types to conserve maximum biodiversity.

Carbon storage is frequently calculated for different vegetation types using an allocated C stock value or remotely sensed indices (e.g. with satellite imagery). Very little field-based research has been carried out over a wide variety of vegetation types and condition states (Watson et al. 2000), hence the effects of plant community composition, age structure and environmental conditions on C storage are largely unknown. The current study was undertaken across a 7100 km^2 study region, in several different vegetation types, and has shown that functional richness of vegetation has a strong influence on the C storage value of plant communities. This relationship was predominantly driven by the addition in functionally rich sites of woody vegetation, which is C-dense compared to herbaceous vegetation.

Erosion mitigation was also influenced by vegetation functional richness, but when river red gum sites were removed, this relationship was no longer evident. In addition to large amounts of organic matter inputs into the soil by river red gum trees (Briggs &

Maher 1983; Cunningham et al. 1992), river red gum sites feature large amounts of litter contributed by floodwaters, which results in both high ground cover and high aggregate stability. River red gum sites were the most structurally rich of all types as they often contained a dense tall shrub layer. As a result of their elevated fertility, river red gum sites were more productive in terms of herbaceous vegetation cover. The combination of high herbaceous cover and high litter cover in river red gum sites was the driving factor in the relationship between erosion mitigation and functional richness.

7.4.3 Ecosystem Service Providers (ESPs) for Bird Conservation, C Storage and Erosion Mitigation Services

Kremen (2005) suggested that the first step in management for ecosystem service provision was the identification of the ecosystem components responsible for service provision, or ecosystem service providers (ESPs). These components can be populations, species, functional groups (guilds), food webs or habitat types. The importance of each component for service provision is defined by its efficacy in providing a particular service and its abundance (Balvanera et al. 2005).

In the current study, cover and biomass of woody vegetation, CWD and litter, cover of herbaceous vegetation, and biomass of dead standing vegetation were important contributors to bird conservation value, C sequestration and erosion mitigation. Cover of all functional groups was positively correlated, except that tall shrub cover was not correlated with herbaceous or litter cover. Biomass of all components was also positively correlated, except that herbaceous biomass was negatively correlated with the other components. Herbaceous biomass was not significantly correlated with provision of any of the three ecosystem services.

The source of litter, CWD and dead standing elements was woody vegetation, hence the strong cover and biomass correlations between these attributes. Woody vegetation was the dominant ecosystem service provider for each of the ecosystem services examined in this study. Herbaceous cover was correlated with woody vegetation cover, explaining the otherwise surprising relationship between woody vegetation and erosion mitigation provision but the relationship between woody vegetation and herbaceous biomass was not significant. A negative relationship between woody and herbaceous biomass is conceivable: dense woody vegetation is a superior competitor for light, the shading effect from woody vegetation resulting in a reduction in herbaceous vegetation biomass (Scholes 2003; Scholes & Archer 1997). In terms of bird conservation value, herbaceous ground cover provided habitat for some species and foraging opportunities for many more species (e.g. perch-and-pounce species including jacky winter and flycatchers), making it an important ESP for this service.

Herbaceous and litter cover was used in the calculation of erosion mitigation value as it is important in protecting the surface soil from raindrop impact. However, biomass production is sporadic in the study region in response to floods or high rainfall, and plant and litter cover is not necessarily present seasonally or year-round. Hence, aggregate stability was considered as important as herbaceous cover. Although herbaceous vegetation is an important ESP for erosion mitigation, litter is equally as important, especially eucalypt leaf litter as it provides litter with a high C:N ratio, which provides a steady resource for decomposer organisms that act to stabilise macro-aggregates (Lavelle & Spain 2001). In terms of C storage, herbaceous vegetation was far less important for C sequestration than woody vegetation and was less important as an ESP in the provision of this service. However, in grassland ecosystems where there is no woody vegetation, herbaceous vegetation was the dominant plant ESP for C storage. The identity and relative importance of ESPs is therefore context-specific and varies among vegetation communities. For example, at the landscape scale, woody vegetation is the dominant ESP for most of the ecosystem services considered here, but at the site scale, this is not necessarily the case.

7.4.4 Vegetation Condition Assessment for Ecosystem Service Provision

An understanding of the ecology underpinning the provision of most ecosystem services is lacking, hindering their management and conservation (Kremen 2005). Information regarding the dominant ESPs for many services is deficient. Hence, what constitutes good or bad vegetation condition for many ecosystem services is largely speculative. To identify the factors influencing the magnitude and variability of ecosystem service provision, Kremen (2005) articulated two complementary approaches: an ESP-centred approach and a function-based approach. The ESP-centred approach focuses on the abundance of an important ESP, and should be used when a functional inventory identifies individual ESPs that are highly uneven in their contribution to ecosystem service provision (e.g. the ecosystem service of interest is provided by a single ESP). On the other hand, if multiple ESPs of roughly similar contribution to service provision exist, the function-centred approach should be used. In the ESP-centred approach, factors affecting the distribution, abundance or efficiency of the ESP are identified, and management is targeted at populations of that ESP. In the function-centred approach, multivariate analyses are used to identify environmental factors influencing aggregate function and its variability in order to identify management priorities.

In the current study, multiple ESPs were identified as contributors to several ecosystem services simultaneously, depending on the context. These ESPs were canopy or woody vegetation, litter and CWD cover and biomass, tall shrub and herbaceous cover, and dead standing biomass. For the ecosystem services examined (i.e. bird conservation value, C storage and erosion mitigation), vegetation condition assessment should consider these ESPs in each specific context or vegetation type, and assess factors affecting their ability to effectively provide services and persist in the landscape. Woody vegetation had a significant influence on the abundance of most other ESPs, most of the ESPs being derived from woody vegetation. Woody vegetation abundance was predominantly related to climate (mean rainfall), soil type, moisture availability (the interaction of climate and soil type), and management (clearing, ring-barking and grazing) (Chapter 2). Where woody vegetation was not present, climate, soil type, moisture availability and management were still important, but short-term climate was more influential (owing to the shorter life span of herbaceous vegetation) and grazing management was more important (discussed below).

The current approach to vegetation condition assessment for biodiversity conservation in Australia (e.g. Gibbons et al. 2008; DSE 2004; Eyre et al. 2008; Croft et al. 2009) is to compare extant vegetation with pre-disturbance benchmarks. By comparing extant vegetation with historical benchmarks, traditional biodiversity conservation assessments automatically downweight the value of ‘degraded’ remnants, particularly in agricultural regions. These remnants may not be iconic sites for biodiversity conservation, but they still have value for ecosystem service provision as was illustrated in the current study. In some instances, they may have enhanced levels of provision of the ecosystem services upon which agriculture relies. These remnants are disproportionately important as they are required to perform the same roles as once contiguous, undisturbed vegetation. Comparison of extant vegetation with benchmarks is unlikely to be useful in terms of vegetation condition assessment for ecosystem service provision for a number of reasons, one being a lack of benchmark data.

Important considerations in the assessment of vegetation condition for ecosystem service provision include (Carpenter et al. 2009; MA 2005): (1) non-linearity between ‘ecosystem intactness’ and ecosystem service provision; (2) complex compensatory responses acting to stabilise service provision as condition changes; (3) different spatial and temporal scales of ecosystem services provision (i.e. ecosystem service provision is not restricted to the remnant under consideration); and (4) spatially and temporally shifting societal values and demands dictating the demand for ecosystem services. Vegetation condition assessment in the context of ecosystem service provision needs to be (1) context-specific, (2) conducted in close consultation with relevant stakeholders and ecosystem service beneficiaries (ESBs; Rounsevell et al. 2010), and (3) constantly adapting according to changes in societal demands. In addition to the purely ecological considerations of traditional vegetation condition assessment, economic and social indicators may also be required for a holistic assessment of vegetation condition for ecosystem service provision.

7.4.5 Influence of Grazing Intensity on Ecosystem Service Provision

The effects of grazing on four ecosystem services (i.e. plant conservation, bird conservation, C storage and erosion mitigation) in river red gum and coolibah communities was studied as these were the only communities where all three grazing intensities were experienced. Grazing significantly affected plant and bird conservation value, and C storage, and all four services were negatively influenced by increasing grazing intensity in these communities. Plant and bird conservation services were not correlated with each other, but increasing grazing intensity had a negative impact on both. Differences in canopy, tall shrub, herbaceous and litter cover, and litter biomass were shown between grazing intensities. However, especially in the case of river red gum communities, it is not possible to conclude cause and effect. Many of the large trees in these communities pre-date settlement and therefore grazing activities in the region, so it is unlikely that grazing activity has resulted in a decline in canopy cover, though grazing may contribute to canopy dieback in river red gum communities through soil compaction and nutrient re-distribution (Reid et al. 2007).

Tall shrub cover may be influenced by grazing. The dominant tall shrubs in river red gum communities were cooba (*Acacia salicina*) and river cooba (*Acacia stenophylla*), both of which are readily browsed by livestock. There is anecdotal evidence to suggest a decline in cooba as a result of livestock grazing (e.g. Allen 1949; Maiden 1889). In coolibah communities, the dominant tall shrubs were river cooba, eurah (*Eremophila bignoniiflora*) and rosewood (*Alectryon oleifolius*), the later two browsed by livestock although eurah is suspected of poisoning stock (Cunningham et al. 1992). The possibility exists that grazing does not occur at sites densely covered by woody vegetation and this is the cause of differences in canopy and tall shrub cover between sites of differing grazing intensity.

In the case of herbaceous vegetation and litter, it is highly likely that increasing grazing intensity negatively affected these vegetation attributes. Grazing disturbs litter dynamics by increasing the rate of decomposition, thereby decreasing both litter cover and biomass (Swift et al. 1979). The consumption of herbaceous biomass also results in decreased litter input, especially from herbaceous sources. The lack of difference in herbaceous biomass between grazing intensities compared to a significant difference in herbaceous vegetation cover is of particular interest here. After examination of the grazing-sensitive and tolerant plant species lists (Chapter 6), it is apparent that there are differences in the growth form of species between the two extremes of grazing intensity. Grazing-sensitive species were often low growing, spreading species, occasionally rhizomatous (e.g. rat's tail couch *Sporobolus mitchellii* and couch grass *Cynodon dactylon*), and often softer, more fleshy species (e.g. wandering jew *Commelina cyanea* and New Zealand spinach *Tetragonia tetragonioides*). Grazing-tolerant species were the opposite: the grasses were tussocky (e.g. native millet *Panicum decompositum* and slender panic *Paspalidium gracile*) and the herbs were often open, subshrub species (e.g. *Sclerolaena* spp. and *Dissocarpus biflorus* var. *cephalocarpus*). This suggests that heavily grazed sites have an uneven distribution of ground cover as they were dominated by taller clumps of dense biomass, whereas rarely grazed sites had a more even distribution of lower biomass plant cover.

Though not demonstrated in this study, the decrease in litter and herbaceous vegetation cover due to grazing should lead to decreased erosion mitigation value of sites as the erosion mitigation score was calculated using a measure of aggregate stability (soil C:N ratio) and ground cover (herbaceous and litter cover). The C:N ratio was correlated with litter biomass, hence all of the attributes used in the calculation of erosion mitigation value were negatively impacted by grazing (herbaceous and litter cover and C:N ratio). It therefore stands to reason that erosion mitigation should be negatively impacted by grazing. The decrease in litter inputs to the soil should also, through time, result in a decrease in soil C storage, and hence the C storage value of the site.

7.4.6 Trade-offs between Grazing and Ecosystem Service Provision

No trade-offs were found between the four services examined in this study since most were positively correlated with each other (Figure 7.3). An increase in the provision of one service should result in the increase of all services. The exception was plant conservation and bird conservation: these services were not significantly correlated. In Chapter 6, of the top 16 sites for plant conservation, only seven were listed in the top 16 sites for bird conservation. This indicates that there are different drivers for plant biodiversity as opposed to bird conservation value. This disparity may be explained by species vs functional group richness. Plant species richness did not correlate with bird conservation value, whereas functional richness in the plant community was significantly correlated with bird conservation value. Where sites were both plant species rich and had high functional richness, they were valuable for both plant and bird species conservation.

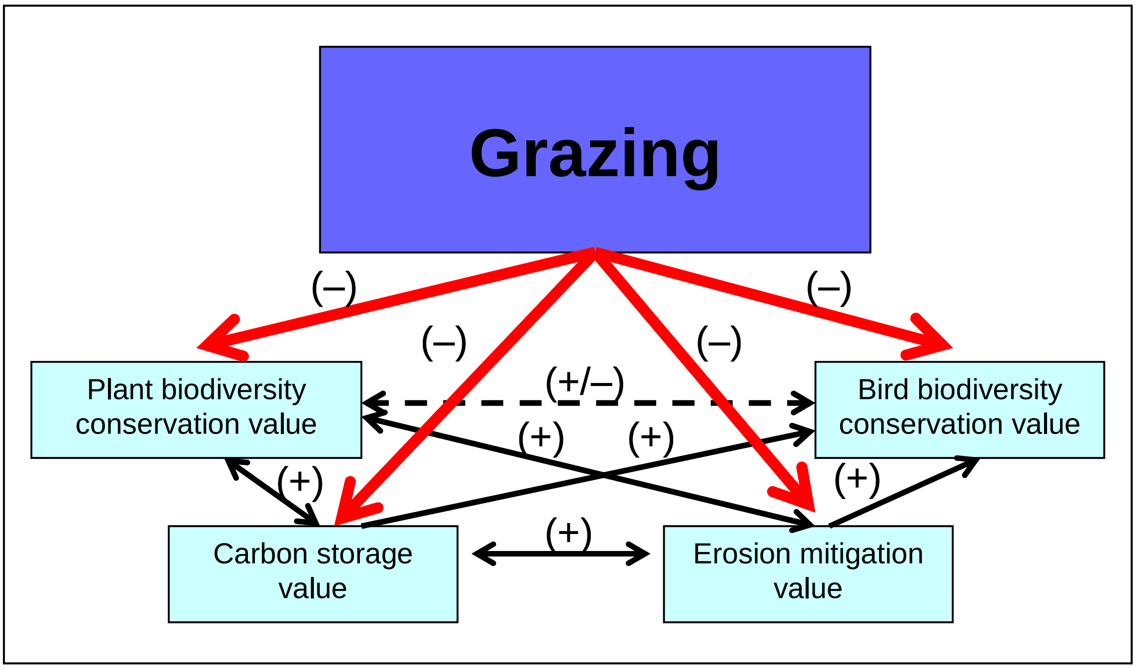


Figure 7.3: Synergies and trade-offs between ecosystem services.

While there were no significant trade-offs between the ecosystem services (Figure 7.3), the lack of correlation between each of the ecosystem services, woody biomass and herbaceous vegetation biomass may present a conundrum for graziers. In order to maximise ecosystem service provision, maximising woody vegetation biomass was highlighted as an attractive management step. However, the negative influence of woody vegetation biomass on herbaceous biomass means that another ecosystem service, forage production, may be reduced, hence resulting in lower livestock production and economic losses for the landholder. Given the negative influence of grazing on plant and bird biodiversity, in particular, and the popularity of grazing as a land use in the study region, this finding is disconcerting. Those areas currently experiencing low grazing intensity need to be protected from the possibility of increased grazing pressure, and serious consideration of current grazing practices in other areas must be considered in order to maximise provision of the services examined here.

7.5 Conclusion

This study demonstrated that functional richness in vegetation communities is more important than species richness in the context of ecosystem service provision (C storage, erosion mitigation and plant and bird conservation). The ESPs for the four ecosystem services should be included in future assessments of vegetation condition in the context of the provision of these ecosystem services. Synergies in multiple ecosystem service provision were highlighted, and potential trade-offs with land use identified. Grazing negatively influenced plant and bird conservation, highlighting an important trade-off between livestock production and nature conservation and C storage. Continued loss of biodiversity as a result of grazing may compromise the ability of remnant vegetation in this agricultural region to provide other ecosystem services. The loss of C storage and sequestering mechanisms may also result in large shifts in ecosystem functioning and service provision in the future, with implications for climate change. Management for ecosystem service provision must strive to meet current land use demands, without compromising the ability of ecosystems to satisfy future demands. Grazing may be depleting the natural capital required for future ecosystem service provision.

Chapter 8: Synthesis and Conclusions

Plains grass (*Austrostipa aristiglumis*) north of Pilliga (May 2008).

Chapter 8: Synthesis and Conclusion

8.1 Introduction

Agriculture relies heavily on a wide range of ecosystem services provided by native vegetation in the landscape (MA 2005; Tschantke et al. 2005; Zhang et al. 2007), hence farmers may want to manage remnant native vegetation on their properties to maximise provision of desired services. Careful management of these areas may attract incentive payments in the future as society recognises the need to support farmers' role in ensuring the persistence of natural capital and the ecosystem services that derive from it (Zhang 2007). This is particularly true for public good services with large-scale benefits (e.g. carbon sequestration, biodiversity conservation and provision of clean water) that extend well beyond the farm gate.

The native vegetation of the lower Namoi floodplain has been extensively cleared and modified by agriculture, either directly through clearing, cultivation, ringbarking and grazing, or indirectly through changes to flood, fire and other disturbance regimes. No areas of native vegetation on the floodplain are part of the formal conservation reserve estate (i.e. the National Reserve System; Commonwealth of Australia 2009) and the large majority is privately-owned and managed. In order to ensure the persistence of remnant vegetation across the study region, landholders need information about its multiple values and the impacts of management decisions on the condition of native vegetation for different uses. This information can then be used in decision making and farm planning processes. Catchment Management Authorities (CMAs) also require this information to guide investment in conservation, restoration and sustainable natural resource management while state and national governments need this information to determine policy, guide investment and meet international obligations (e.g. for biodiversity conservation and greenhouse gas emissions).

Over the last decade, the Australian cotton industry through its Best Management Practice (BMP) initiative has been proactive in seeking information about ecosystem services provided by natural areas on farms and management to maximise ecosystem service provision. Cotton production on the lower Namoi floodplain occupies approximately 10% of the land area, but natural areas on cotton farms comprise the full spectrum of landscape positions and soil types, from frequently inundated riparian zones on silty alluvial soils, to regularly inundated floodplains and secondary floodplains on fertile, heavy cracking clay soils, to sandy, sodic ridges that are rarely flooded. This generates a high degree of natural variability in the native vegetation. In addition, management of native vegetation on cotton farms varies. Some landholders choose to conserve or rarely use natural areas for agricultural purposes on their properties, while others graze sheep and cattle at varying densities. Some areas have been thinned or cleared in the past and allowed to regenerate, while other areas that were once wooded now exist as derived grasslands.

The research reported in this thesis is important. There are few records of what vegetation exists on the lower Namoi floodplain, especially on private land. Much has been published as a result of extensive round-table collaborative efforts on the provision of various ecosystem services in different ecosystems, but most use coarse datasets (e.g. Costanza et al. 1997; Naidoo et al. 2008). By contrast, little research has quantified multiple ecosystem service provision in contrasting vegetation types in the field, at a scale that is useful for individual landholders, for most ecosystems of the world. Few studies have investigated how management influences ecosystem service provision and the trade-offs in ecosystem service provision associated with varying land use intensities, though numerous calls have been made for such information (Bennett et al. 2009; Carpenter et al. 2006; DeFries et al. 2004; Farber et al. 2006; Kareiva et al. 2007; Nelson et al. 2009; Rodríguez et al. 2006). Recommendations for the management of vegetation for multiple ecosystem service provision are therefore lacking.

8.1.1 Aims and Objectives

The preceding chapters of this thesis have:

1. Defined common vegetation communities of the lower Namoi floodplain;
2. Mapped the distribution of woody vegetation across a 7000 km^2 area of the floodplain;
3. Quantified the provision of three ecosystem services (carbon sequestration, erosion mitigation and biodiversity conservation) in five vegetation communities commonly encountered on the lower Namoi floodplain; and
4. Investigated the relative contribution of species and functional richness to ecosystem service provision, identified ecosystem service providers (ESPs) for three ecosystem services, and described management related trade-offs associated with provision of these three ecosystem services.

This chapter provides a synthesis of this research and discusses how it contributes to theory and knowledge of ecosystems and the services they provide. Limitations of the research presented in previous chapters are discussed and management recommendations for maximising ecosystem service provision are provided. Future research needs in the context of quantifying multiple ecosystem service provision and management of vegetation for maximum service provision are also suggested.

8.2 Summary of Main Findings

8.2.1 Vegetation Communities of the Lower Namoi Floodplain

Differences in soil properties, flood frequency and distribution, and annual rainfall create conditions that differentially favour river red gum (*Eucalyptus camaldulensis*), coolibah (*E. coolabah*), myall (*Acacia pendula*), black box (*E. largiflorens*) and grassland communities, and influence the dominance of different life-forms in each community. Moisture availability is dictated by climate, flood inundation patterns and soil type (i.e. EC levels and soil texture). Management (i.e. clearing history, ringbarking, grazing and altered flood patterns) influences the occurrence of vegetation communities as well, but Chapter 2 reported patterns in the distribution of long-lived, native species to minimise confounding by short-term management effects.

River red gum riparian forests were found on silty soils with high N, organic C and Ca fringing the Namoi River channel. Two structural variants of this community were present, one with a grassy understorey, the other with an *Acacia*-dominated shrub layer. The understorey was characterised by a dominance of forbs and graminoids, annual grasses and short-lived perennial species, and high abundance of exotic species such as lippia (*Phyla canescens*). Coolibah woodland and open-woodland sites were located on the heaviest clay soils, usually with high ECEC, Mg, Ca, and pH. The understorey was dominated by perennial grasses, though sites on minor drainage lines experiencing more frequent inundation had greater canopy cover and a sparse ground layer dominated by forbs. Some sites also had a dense cover of chenopods as opposed to grasses.

Myall tall shrubland and tall open-shrubland sites occurred predominantly on sandier, sodic soils characteristic of prior stream formations. These sites were not frequently flooded so the understorey vegetation was dominated by chenopod species and less flood-tolerant grass species. The soil surface was often crusted, impeding infiltration, and hence contributing to the dominance of chenopod species and an open understorey layer with large areas of bare ground, even in good seasons. Black box open-woodlands were commonly observed on lesser inundated floodplains with similar soil characteristics to the myall group, that is sodic, ‘sandy’ soils. Again the understorey was dominated by sparse chenopod shrubs, occasionally with a tall shrub layer dominated by eurah (*Eremophila bignoniiflora*). This vegetation community only occurred in the western end of the study region west of Burren Junction.

Grasslands occurred across all soil types and inundation frequencies. Several were derived from previously wooded areas that had been cleared or intensively ring-barked and had a different species composition to ‘true’ (natural) grasslands. Several grassland communities were distinguished, these compositional differences being predominantly due to moisture and soil physico-chemical differences. Grasslands on the least frequently flooded soils were dominated by Mitchell grass (*Astrebla lappacea*), while frequently flooded sites were dominated by Warrego summer grass (*Paspalidium jubiflorum*), or plains grass (*Austrostipa aristiglumis*). Grassland on very heavy clay soils was dominated by coolibah grass (*Panicum queenslandicum* var. *queenslandicum*) with *Leptochloa divaricatissima* co-dominant. This grassland community has previously been undescribed in other studies (e.g. Benson 2006) and may be unique to the lower Namoi floodplain between Wee Waa, Burren Junction and Rowena, although its derivation is unknown.

The number of sites in each vegetation type was unbalanced as some vegetation types, such as myall and black box were not common across the study region. In the case of myall sites, representation was fairly even across the region, but black box sites were restricted to two properties in the western end of the study region. The magnitude of management-related impacts at these two properties (e.g. long grazing history and intensive ring-barking) may have biased the data on the composition of black box vegetation.

8.2.2 Woody Vegetation Mapping

Individual vegetation communities could not be mapped using single-date SPOT 5 imagery, but woody vegetation across the lower Namoi floodplain was mapped with 85% accuracy. Mapping was carried out using unsupervised classification of ten SPOT 5 scenes captured in summer 2004–05. Unsupervised classification using 50 classes gave similar results to the 100 or 200 classes typically used in similar studies (e.g. Barson et al. 2000), hence similar accuracy was achieved with cheaper, less resource and labour-intensive methods. This study indicated that approximately 7% of the lower Namoi floodplain is covered by vegetation dominated by woody plants, the majority of which occurs along drainage lines.

The imagery used in this study was not captured during a time period conducive to vegetation mapping but was contributed in-kind to this project and was being used by state agencies at the time. Widespread rainfall prior to and during the image capture period resulted in a pulse of herbaceous vegetation growth which was then confused with woody vegetation by the unsupervised classification. Vigorously growing crops may also have been confused for woody vegetation. These two factors may have led to an overestimation of the extent of woody vegetation across the region. On the other hand, the spectral characteristics of some vegetation types, in particular, myall, which typically have low reflectance in the NIR wavelengths used to distinguish vegetation, may have resulted in an underestimate of woody vegetation cover.

8.2.3 Carbon Storage Value of Lower Namoi Floodplain Vegetation

Carbon storage was measured at 60 sites across the five vegetation types and included C stored in the woody, herbaceous and dead standing components, coarse woody debris (CWD), litter, roots and soil components at each site. The C storage value of vegetation and top soil (0–30 cm) across the lower Namoi floodplain was highly variable, ranging from 22 t C ha^{-1} in a grassland site near Walgett to 396 t C ha^{-1} in a river red gum site east of Narrabri. River red gum sites averaged 216 t C ha^{-1}, 101 t C ha^{-1} in coolibah, 70 t C ha^{-1} in myall, 52 t C ha^{-1} in black box and 40 t C ha^{-1} in grasslands. The majority of the C was stored in woody biomass, though soil C made up a large proportion of the C in grassland vegetation. Carbon stored in all ecosystem components except herbaceous vegetation was positively correlated with woody vegetation. River red gum vegetation was the most valuable for carbon storage, in particular the woody biomass component.

Allometric equations were used to estimate woody vegetation biomass. This method (e.g. Snowdon et al. 2000) introduces inaccuracies in carbon store estimates for woody vegetation. Allometric equations developed by the Australian Greenhouse Office (AGO) were developed for a range of species over a wide climatic range by growth form; equations were not species-specific. The AGO equations use diameter at breast height (DBH), diameter at 30 cm above the ground, or height only to estimate total tree biomass. Large differences in wood density are known to occur between species examined in the current study. For example, differences in wood density alone are likely to result in large differences in the woody biomass of individual trees, and this error is multiplicative with each additional tree. There are also likely to be differences in the rate of hollow development between species. These factors are not included in the calculation of tree biomass and need greater attention to make the C storage estimates of woody vegetation more accurate.

There were limitations in the estimation of soil C storage. Vertosols are known to shrink and swell with variation in soil water content. This produces large differences in soil bulk density and the volume of soil sampled when sampling to a fixed depth. Bulk density was measured using 85 mm soil cores as required by the Australian standard (McKenzie et al. 2000), but in vertosol soils this sized core may not give an accurate measure of bulk density due to the size and distribution of cracks through the soil. Consideration should be given in future studies to measuring soil bulk density in vertosol soils using a larger diameter soil core separate to C sampling cores, in order to obtain a more accurate measurement of C storage.

8.2.4 Erosion Mitigation Value of Lower Namoi Floodplain Vegetation

Macro and microaggregate stability were measured at 60 sites across the five vegetation types on the lower Namoi floodplain at two depths, 0–5 cm and 20–30 cm. It was considered important to measure these two depths as the cracking nature of the heavy clay soil frequently results in subsurface soil being exposed to large amounts of water, especially after extended dry periods. The mean weight diameter of aggregates after wet sieving (Kemper & Rosenau 1986) was used as the measure of macroaggregate stability, while the ASWAT test (Field et al. 1997) was used to assess microaggregate stability (clay dispersion) after rapid wetting. Mean weight diameter of aggregates (macroaggregate stability) in the surface soil was positively related to soil C:N ratio and a linear regression model adequately described the relationship. In subsurface soils, the relationship was more complex. Multiple linear regression revealed that organic C, C:N ratio, K, Ca and exchangeable Mg were important contributors to macroaggregate stability in subsurface soils. Woody vegetation cover and litter cover were the most important components of the vegetation in dictating surface soil C:N ratio. The relationship between ASWAT score (microaggregate stability) and soil physico-chemical characteristics was complex in surface soils, strongly significant correlations being found between Na and ESP (promoting clay dispersion), and Ca and exchangeable Ca (reducing clay dispersion). Silt, Ca:Mg and C:N also acted to reduce clay dispersion in the surface soil but did not produce as strong correlations and, were considered less important. In the subsurface soil, Na was the strongest determinant of clay dispersion, soils with high Na being more dispersive.

In order to determine the value of vegetation for erosion mitigation, soil C:N and ground cover were used to rank sites according to the two variables equally. River red gum sites were the most valuable for erosion mitigation on account of their high C:N ratio and combined herbaceous and litter cover. These two variables addressed erosion by slow moving floodwater (in which macroaggregate stability was considered to be the more important soil property) and raindrop impact (where cover was considered more important). Raindrop impact was not of great concern in the majority of the soils of the lower Namoi floodplain owing to their self-mulching surface. Aggregate stability was a greater concern as the soils have a high propensity to slake and disperse, allowing them to be easily eroded by floodwaters.

8.2.5 Biodiversity Conservation Value of Lower Namoi Floodplain Vegetation

Vascular plant and bird biodiversity were investigated at 54 sites across the lower Namoi floodplain in the five vegetation types. Native plant species richness did not differ significantly between vegetation types, but rare species were more abundant at river red gum sites. Most plant species were recorded at only one to five sites. Three species (black rolypoly *Sclerolaena muricata*, tarvine *Boerhavia dominii* and creeping saltbush *Einadia nutans*) were recorded at most sites. In terms of vascular plant conservation value, river red gum sites ranked highest on account of their abundance of rare species.

River red gum and coolibah sites contained the greatest number of bird species of the five vegetation types (15 and 13 respectively). No differences were found between vegetation types for bird rarity rankings or bird conservation value. This was due to habitat requirements of different bird species, i.e. some species were confined to certain habitat types while others moved between habitat types. Most bird species were recorded in four or less sites across the 54 sites. Considering both plant and bird conservation value together, only seven sites were for both plant and bird conservation value, ranking in the top 30% (16 of 54 sites) for both services.

8.2.6 Congruence between Plant Species Richness and Functional Richness and Ecosystem Service Provision

The ecosystem services examined (carbon storage, erosion mitigation, plant conservation and bird conservation) were all positively correlated except plant conservation and bird conservation, which were not significantly correlated. Plant species richness showed little correlation with ecosystem service provision, but plant functional richness was positively correlated with all ecosystem services. River red gum sites were the most functionally rich while grasslands were the least functionally rich. Plant functional richness was important for bird conservation as more habitat structure, nesting, perching and foraging opportunities, and protection was provided in structurally complex vegetation. Greater functional richness was also beneficial for carbon storage, greater functional richness being equated with higher plant production, especially woody biomass. The relationship between functional richness and erosion mitigation was driven by river red gum vegetation, which was both functionally rich and highly productive in terms of input of high C:N litter and herbaceous groundcover, the two variables important for erosion mitigation value. When river red gum sites were excluded from the analysis, the relationship between functional richness and erosion mitigation was no longer significant.

8.2.7 Ecosystem Service Providers Responsible for Generation of Ecosystem Services

The components of ecosystems that generate ecosystem services are termed ‘ecosystem service providers’ (ESPs, Kremen 2005) or simply ‘service providers’ (Luck et al. 2009). ESPs important in generating C storage, erosion mitigation and bird conservation services included trees, dead standing wood, CWD, tall shrubs, litter and herbaceous vegetation. Biomass of these components was especially important for most services, although ground cover was assumed to be important for erosion mitigation. All of the ESPs were positively correlated with woody vegetation biomass except herbaceous vegetation biomass, which was negatively correlated with woody biomass. Measurements of the abundance of these ESPs, and abiotic elements (e.g. soil properties and moisture availability) and management influencing their abundance and efficacy in providing ecosystem services (e.g. grazing intensity, clearing or thinning) should be included in vegetation condition assessments in the context of these ecosystem services.

8.2.8 Impact of Grazing on Ecosystem Service Provision

Grazing affected plant and bird conservation values through its negative influence on rare species. Carbon storage was also affected by livestock grazing as biomass is removed by grazing animals. Grazing may have influenced woody vegetation by encouraging tree canopy dieback (Reid et al. 2007) and altering recruitment patterns (Dorrough & Moxham 2005). Livestock remove herbaceous cover and disturb litter decomposition processes (Swift et al. 1979), reducing aggregate stability so the lack of a significant result between grazing and erosion mitigation was surprising. However, there was a negative trend in the data. Two possible explanations for the lack of difference between grazing intensities in provision of erosion mitigation were: (a) soils are more resilient to erosion as a result of inherent ecosystem properties (e.g. high CWD input to the soil by aging trees), or (b) grazing is mining soil carbon reserves but the critical threshold for erosion mitigation service provision has not been crossed.

Grazing-tolerant and sensitive species of vascular plants and birds in river red gum and coolibah sites were determined according to their abundance in regularly and rarely grazed sites. Plant species tolerant of grazing were generally unpalatable, increaser or early successional species, often with wind dispersed seed. Grasses were tussocky and herbs were often almost woody. Grazing-sensitive species were generally palatable or remained green when other pasture species had become rank through the growing season, were low growing and softer or fleshier species susceptible to trampling by hard-hooved livestock. Grazing-sensitive bird species were predominantly ground or canopy feeding, insectivorous, woodland specialists. Grazing tolerant bird species were mostly omnivorous hollow nesters preferring open-woodland habitats. Grazing negatively impacts ground foraging potential for birds through biomass removal and trampling. Grazing may also lead to tree canopy dieback (Reid et al. 2007), hence negatively impacting ground and canopy feeding birds. However, senescent and dieback affected trees provide habitat for hollow nesting birds, and those species not reliant on a single food source (i.e. omnivores) are able to persist in grazing landscapes.

8.3 Synthesis

A conceptual diagram was developed to illustrate the dynamic interactions between climate, abiotic and social influences, ESPs and ecosystem service provision

(Figure 8.1). This study concentrated on the ecological aspects of ecosystem service provision, but the influences of policy, society and individual landholder values on ecosystem service provision were also observed and are included in the diagram for completeness.

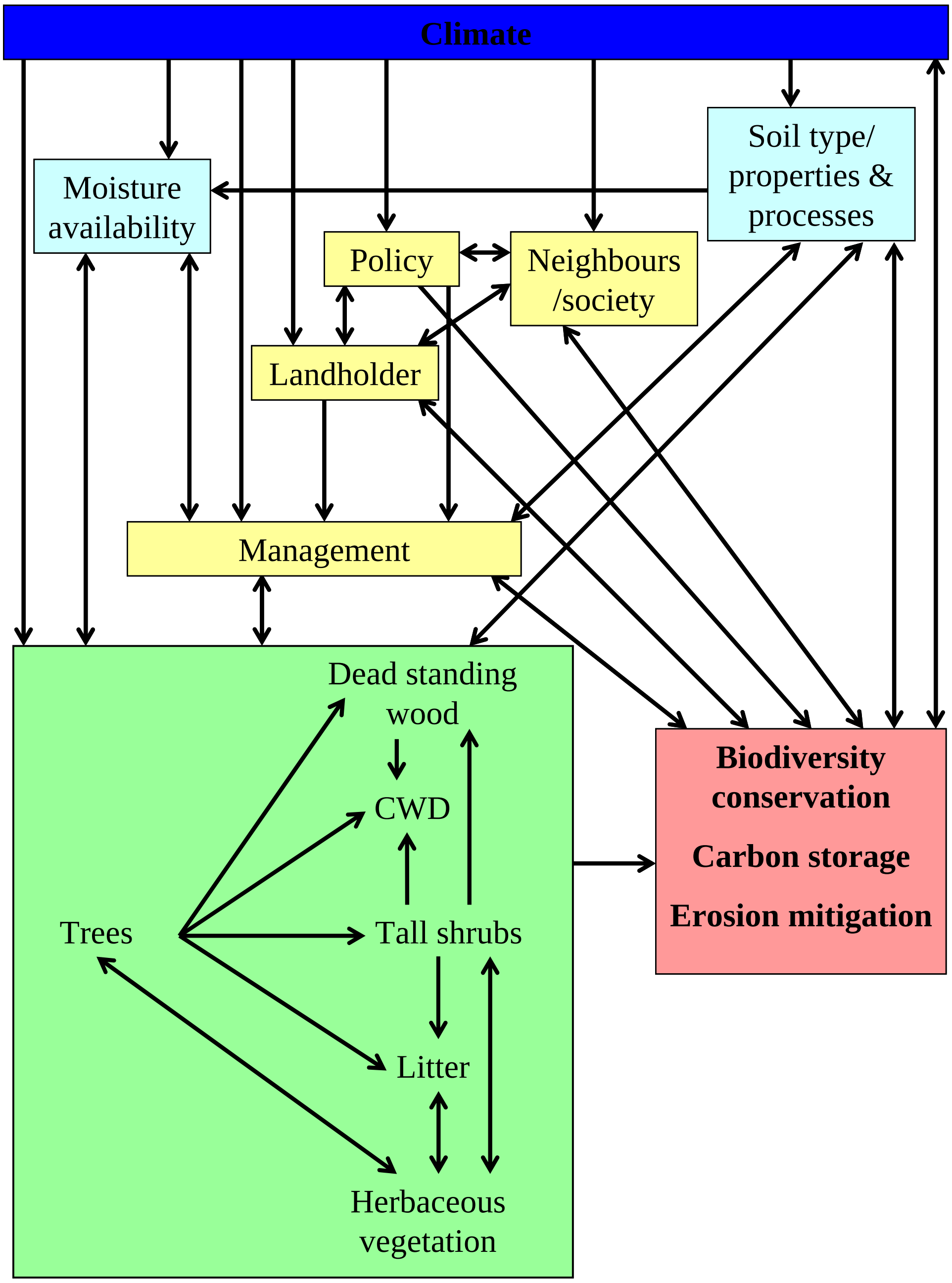


Figure 8.1: Conceptual diagram of the relationships between abiotic and societal influences on ESPs and provision of three ecosystem services.

Climate is the overriding influence on all elements in the diagram. Long-term climate influences soil type and its properties and processes, and the presence of ESPs (i.e. long-lived components such as large trees), while short-term climate, directly and in cooperation with soil factors, determines moisture availability and hence vegetative growth (abundance of ESPs). This was demonstrated early in the study through the effects of drought followed by flooding across the region (Figure 8.2). Soil texture and chemical properties also influenced moisture availability, with high-EC soils being more water limiting environments due to high osmotic potential. Ecosystem service providers, in turn, influenced moisture availability and soil properties and processes by providing litter inputs to the soil and regulating microclimate through shade and temperature effects at the soil surface.

Figure 8.2: Changes in vegetation cover and biomass in a grassland site north-east of Pilliga during drought (spring 2007) and after above-average rainfall (spring 2008).

While not investigated, changes in climate influence societal and community values and policy. In the current study, the issue of water shortage arose and water supplies, both surface and subsurface, needed to be prioritised. Water was often diverted from irrigation uses in this situation. Climate influenced landholders' economic situation and hence their values, although their management is bound by policy such as vegetation regulations. Society, community and neighbours influence management through policy and by lobbying individual landholders. The effects of wider society and neighbours and their push for responsible environmental management (e.g. fencing livestock from riverbanks and management of stock routes) was observed during this study. The impacts of landscape-scale management on ecosystem service provision was not recorded in the current study, but water diversion and levee development, as well as the hostility of adjacent land uses for various organisms likely affected ecosystem service provision, especially conservation value.

Management of non-cropping areas was influenced by a number of factors including climate and moisture availability, individual landholders' aims and goals, policy, and soil type, properties and processes. Soil properties ultimately dictated the response of the vegetation to rainfall and high moisture availability. On sodic soils where the soil surface had formed a crust, it is likely that infiltration was reduced; hence, the vegetation was unable to respond. Herbaceous biomass on the sodic myall and black box soils was always lower than sites on self-mulching soils. When soil moisture availability was high after rainfall, management was geared to take advantage of the better conditions. Some landholders waited longer than others to increase management intensity based on their economic situation and attitudes and perceptions towards vegetation resources. Some landholders could afford to spell paddocks for multiple seasons and some made it a priority where they perceived management during drought conditions had degraded the resource. Landholder management and the interactions between management, climate and soil were the driving factors determining ESP abundance and efficacy in providing ecosystem services.

Of the ESPs identified through this study, trees were the most important. Trees were the main drivers of carbon storage and the erosion mitigation value of vegetation, and also played a large part in the bird conservation value of sites. Trees influenced the abundance of all other ESPs either directly, for example, through the provision of litter, CWD and dead standing wood, or indirectly through competition for resources in the case of herbaceous biomass, which was negatively correlated with woody biomass.

In addition to ESPs, climate and soil properties influence ecosystem service provision. Climate influences bird breeding and hence biodiversity conservation, rainfall patterns influence the need for erosion mitigation services, and temperature influences decomposition rates and other ecosystem processes that determine C storage value. Ecosystem services also influence climate. For example, there is evidence that C pollution is affecting global climate (e.g. Crowley 2000; Houghton 1999), hence C storage may be part of the solution in regulating global climate. As the ecosystem service concept is built around society and human well-being, society decides which ecosystem services are important and hence dictates management directly, as well as indirectly through policy designed to regulate ecosystem service provision.

8.4 How the Research Adds to Knowledge

Ecosystem service theory has been largely speculative to date with little field-based validation of hypotheses or concepts. Being a new field of research, the ecosystem services paradigm is in its infancy, although support for the concept is widespread. This study was predominantly field-based and provides biophysical data describing multiple ecosystem service provision by different vegetation types, and trade-offs with increasing land use intensity. This project spanned a large region with sites ranging across soil, rainfall, flood frequency and management gradients. Most studies to date that have investigated multiple ecosystem service provision have done so over a narrow range of environmental conditions (e.g. Maass et al. 2005).

In a more localised context, the vegetation survey data collected during this study builds on existing knowledge of the distribution of vegetation communities across the lower Namoi floodplain, specifically on private land. Few survey data have been collected previously on private land in the region owing to landowner sensitivities towards government policy and vegetation management. Little information is available on extant vegetation on private land, except where data have been collected remotely. Existing data are based on surveys from travelling stock routes and other public lands which are managed differently to private lands (Davidson et al. 2005).

8.5 How Research can contribute to Resource Management Practice

The ecosystem services concept is potentially attractive to a broad field of land managers as it integrates the biophysical, economic and social aspects of environmental management. The ecosystem services concept incorporates human activities as an integral component of ecosystems rather than as a regulating force over them. In productive arable lands such as the lower Namoi floodplain, responsible management of native vegetation can only be achieved on a large scale when landholders realise the full value of their natural capital and understand the positive and negative consequences of their management in relation to each service. Many landholders feel a moral obligation to responsibly manage native vegetation on their properties and want to leave it in better condition than when they acquired it. One landholder remarked that ‘we don’t own the land; we are just looking after it for the next generation.’ Most landholders have good intentions, but often don’t have the knowledge to inform their decision making or the resources to effect decisions if they do. The current study provides landholders with detailed understanding of some of the values of their native vegetation for multiple ecosystem service provision and advice about the effects of management on these values.

Information collected during this study will be incorporated into the cotton industry’s BMP, which includes a module on land and water management. To date, the BMP has developed endpoints and goals to guide growers, but little supporting information on how to achieve management goals has been produced due to the lack of research on native vegetation in cotton growing regions. This research will provide information to help growers identify vegetation communities on their properties and guide management in the context of multiple ecosystem service provision.

This research also provides valuable information for catchment management groups on the vegetation communities and biodiversity of the lower Namoi floodplain and some of the characteristics of these communities. Vegetation management is regulated in NSW through the *Native Vegetation Act 2003* and the *Native Vegetation Regulation 2005*. Benchmark information on the presence and abundance of different life forms are used to assess vegetation condition in the context of management through the *Native Vegetation Regulation 2005*, which ultimately dictates how landholders can manage vegetation. For some communities, benchmark data for assessment of vegetation condition are incomplete or inaccurate. This research will be able to update available information and provide more accurate benchmarks.

8.6 Management Recommendations

The most limiting resource across the lower Namoi floodplain is water. Any practice that alters the availability and distribution of this resource in space or time will generate trade-offs. The management of water needs to be considered at a range of scales. At the landscape scale, floodwater diversion and extraction of groundwater affect entire ecosystems. At a much finer scale (i.e. within individual soil aggregates), moisture availability affects decomposition and nutrient cycling processes. In a temporal sense, the availability of water at key times in the vegetative growing season has the potential to influence vegetation community composition and biomass production. Water management in both space and time has the potential to influence ESPs and hence ecosystem service provision.

At the landscape scale, individual landholders contribute to water movement patterns, and the cumulative impact of development across the catchment is large. The development of irrigation infrastructure across the floodplain has led to altered flood movement and intensity, and hence changes in the distribution of water availability in space and time. At the paddock scale, removal of vegetative biomass and cover, whether it be woody, herbaceous, litter, coarse woody debris or other, has the potential to alter soil water availability. Soil water content in vertosol soils strongly influences their cracking and churning behaviour. This process is moderately resilient as it is related to the inherent physico-chemical characteristics of the soil. However, the degree of cracking and churning can be impacted by land management practices, and this has flow-on consequences for infiltration and runoff, and also microtopographic variation at the soil surface. Gilgai microtopography is generated through the cracking and churning of the soil, hence disruption to this process can influence water retention.

Intense grazing has differentially affected both the plant and bird conservation value of sites through the loss of rare species. Regularly grazed sites across the lower Namoi floodplain are stocked for the majority of the year in large paddocks. Alteration to current grazing practices may minimise this trade-off, but more research is needed. To date, there has been little research on ‘cell’ and other planned grazing strategies in the western half of the state. By grazing paddocks with more stock for shorter periods of time, the vegetation is allowed to recover, and livestock are less able to selectively graze favoured species, resulting in a more even distribution of grazing pressure across all species in the sward.

Increasing grazing intensity also reduced the C storage value of the vegetation through herbaceous vegetation removal. Indirectly, grazing influenced C storage value as associated management practices such as ‘cleaning up’ (i.e. removal of fallen timber) and management of woody vegetation regrowth common practice to aid in ease of stock mustering. Removal of any vegetative biomass, live or dead, reduces the C storage value of the site.

8.7 Future Research

8.7.1 Vegetation Community Distribution

Further research is required to determine the influence of flooding and groundwater on vegetation communities and ecosystem service provision. Moisture availability was the critical influence on plant community composition and distribution, but more detailed information is required to verify the dependence of terrestrial vegetation communities on surface and groundwater and impacts of diverting surface and subsurface water on vegetation and ecosystem service provision. In addition to grazing management, water management is likely to alter the persistence of the full complement of biodiversity in the landscape and hence continued supply of ecosystem services to agriculture in the future.

A more systematic flora survey across the lower Namoi floodplain is required to fully assess the distribution of different plant communities and associated soil types. Site selection in the current study was made opportunistically, although every effort was made to include sites across the full spectrum of compositional and structural variation in each vegetation type. However, some vegetation types such as myall and black box were under-represented due to lack of access and the narrow range of environmental conditions required by these communities.

8.7.2 Vegetation Mapping

The current study found that vegetation community mapping from single-date SPOT 5 imagery was not possible. The use of multiple image dates was limited by cost and time constraints. However, the use of multiple dates could lead to more accurate mapping of vegetation communities due to differences in plant phenology and spectral signatures through time. Image capture during April or May might avoid the confusion of woody vegetation with vigorously growing crops, but image capture needs to be timed in relation to rainfall. April and May are typically low rainfall months in the study region, most cotton crops are being harvested and most wheat crops are just being planted at this time therefore minimising the influence of these land cover types on mapping projects. The use of supervised classification techniques should also be investigated.

8.7.3 Carbon Storage

The accuracy of allometric equations for species on the lower Namoi floodplain should be investigated to reduce errors in woody biomass estimates and produce more accurate assessments of C storage by native vegetation. Development of species-specific allometric equations would greatly enhance the accuracy of woody biomass estimation. Equations should be based on all age and size classes in order to accurately account for hollow development and tree form.

Information on soil processes, such as decomposition rates and the effects of management on these processes, would contribute to a better understanding of the management of soil C storage. Grazing affects soil processes, including decomposition, but the magnitude of these effects on vertosol soils outside temperate regions is inadequately understood. In addition to C storage, decomposition processes are important in aggregate stabilisation and nutrient cycling.

8.7.4 Erosion Mitigation

Erosion mitigation value is a complex ecosystem service to measure. The best way to measure erosion mitigation value would be through measurement of runoff and sediment production in runoff plots or at paddock scale. In the current study, erosion mitigation value of vegetation was inferred from aggregate stability and ground cover. This substituted for field measurement of runoff but provided an indirect measure of erosion mitigation value. It was not possible to use rainfall simulation due to prolonged drought during the early part of the current study and the extensive development of soil cracks. Time to first runoff during rainfall simulation would have would have been in the order of hours as no runoff would have been achieved prior to closure of soil cracks. The amount of water required to achieve runoff would have been impractical. Given the right seasonal conditions, measurement of runoff and sediment production in different vegetation types could supplement the current study.

Infiltration rates were not investigated in this study, but Taddesse et al. (2002) suggested that the impacts of domestic livestock on soil properties and infiltration may result in greater runoff, increasing the risk of erosion. Measurement of infiltration on vertosol soils depends on initial water content so it may be necessary to pre-wet the soil in order to remove this influence on infiltration. Nevertheless, measurement of infiltration may provide greater insight into the effects of grazing management on erosion mitigation.

8.7.5 Biodiversity Conservation

The current study investigated vascular plant conservation value from two seasons of data in a year after a prolonged drought. Survey periods were timed after abundant rainfall and as a result of the drought, livestock numbers were at a minimum. Hence, the results indicated the longer-term effects of grazing without being confounded by current livestock grazing. More sample periods over successive seasons could track the longer term recovery of vegetation after drought.

Bird censuses were carried out over a 2-week period in September 2007. Drought conditions continued during this period and probably affected the presence and abundance of many bird species. Repeat sampling now that the drought conditions have been relieved may provide additional insights. The effect of an environmental flow down the Namoi River during the census period may also have influenced survey data along the river.

8.7.6 Functional Richness Impacts on Ecosystem Service Provision

A limited number of functional groups was used in this study to assess the impact of functional richness on ecosystem service provision, but robust relationships were demonstrated nonetheless. More detailed classification of plant functional groups and different classification schemes could yield additional information. For example, inclusion of plant attributes such as life cycle (annual vs perennial) could affect carbon sequestration and erosion mitigation values, while phenology variables including timing of flowering and seed set could influence bird conservation value.

8.7.7 Other Ecosystem Services

This study concentrated on only three of the many ecosystem services provided by native vegetation. The response of other services to land use intensity may be different to that documented here, and warrants further investigation. There may also be trade-offs between other ecosystem services in these vegetation communities. Documentation of the provision of many more services will help landholders gain a greater understanding of the full value of native vegetation on their properties and the effects of management on those values.

References

Ahmed N. (1996) Occurrence and distribution of vertosols. In: *Vertosols and Technologies for their Management* (eds N. Ahmed and A. Mermut). Elsevier, Amsterdam.

Allen G. H. (1949) Notes on the plants of south-western Queensland. *Mimeo,* 83.

Amezketa E. (1999) Soil aggregate stability: A review. *Journal of Sustainable Agriculture* **14**, 83-151.

Analytical Software. (2003) Statistix 8 User's Manual. Analytical Software, Tallahassee.

Anderson M. J., Gorley R. N. & Clarke K. R. (2008) *PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods*. PRIMER-E Ltd, Plymouth, United Kingdom.

Andreasen J. K., O'Neill R. V., Noss R. & Slosser N. C. (2001) Considerations for the development of a terrestrial index of ecological integrity. *Ecological Indicators* **1**, 21-35.

Annapurna K., Tilak K. V. B. R. & Mukerji K. G. (1996) Arbuscular mycorrhizal symbiosis recognition and specificiy. In: *Handbook of Vegetation Science: Concepts in Mycorrhizal Research* (eds H. Lieth and K. G. Mukerji). Kluwer Academic Publishers, Dordrecht.

Anon. (1995) Sustaining the world's forests: The Santiago agreement. *Journal of Forestry* **93**, 18-21.

Anon. (2007) Criteria and indicators for the conservation and sustainable management of temperate and boreal forests, The Montréal Process.

Apan A. A., Raine S. R. & Paterson M. S. (2002) Mapping and analysis of changes in the riparian landscape structure of the Lockyer Valley catchment, Queensland, Australia. *Landscape and Urban Planning* **59**, 43-57.

Aronson J., Milton S. J. & Blignaut J. N. (2007) Restoring natural capital: Definitions and rationale. In: *Restoring Natural Capital: Science, Business, and Practice* (eds J. Aronson, S. J. Milton and J. N. Blignaut). Island Press, Washington, DC.

Ayers D., Seddon J., Briggs S., Doyle S. & Gibbons P. (2005) Interim benchmarks for the BioMetric tool. New South Wales Department of Environment and Climate Change, Canberra.

Bacon P. E., Stone C., Binns D. L., Leslie D. J. & Edwards D. W. (1993) Relationships between water availability and *Eucalyptus camaldulensis* growth in a riparian forest. *Journal of Hydrology* **150**, 541-61.

Baker A. (2009) Sub Surface Sodicity in the Namoi Catchment. Namoi Catchment Management Authority, Tamworth, Australia.

Baldock J. A. & Skjemstad J. O. (1999) Soil organic carbon/soil organic matter. In: *Soil Analysis: An Interpretation Manual* (eds K. I. Peverill, L. A. Sparrow and D. J. Reuter), Collingwood.

Balmford A., Bruner A., Cooper P., Costanza R., Farber S., Green R. E., Jenkins M., Jefferiss P., Jessamy V., Madden J., Munro K., Myers N., Naeem S., Paavola J., Rayment M., Rosendo S., Roughgarden J., Trumper K. & Turner R. K. (2002) Economic reasons for conserving wild nature. *Science* **297**, 950-3.

Balvanera P., Daily G., Ehrlich P., Ricketts T., Bailey S., Kark S., Kremen C. & Pereira H. (2001) Conserving biodiversity and ecosystem services. *Science* **291**, 2047.

Balvanera P., Kremen C. & Martínez-Ramos M. (2005) Applying community structure analysis to ecosystem function: examples from pollination and carbon storage. *Ecological Applications* **15**, 360-75.

Bange M., Constable G., McRae D. & Roth G. (2010) Cotton. In: *Adapting Agriculture to Climate Change* (eds C. Stokes and M. Howden). CSIRO publishing, Canberra.

Banks R. G. (1995) *Soil Landscapes of the Curlewis 1:100 000 Sheet Report*. Department of Conservation and Land Management, Sydney.

Banks V. (2006) An Investigation of Eucalypt Dieback on the Northern Liverpool Plains, NSW. University of New England, Armidale.

Barker S. & Lange R. T. (1969) Effects of moderate sheep stocking on plant populations of a black oak–bluebush associations. *Australian Journal of Botany* **17**, 527-37.

Barrett G. & Silcocks A. (2002) Comparison of the first and second atlas of Australian birds to determine the conversation status of woodland-dependent and other bird species in New South Wales over the last 20 years. (ed Birds Australia). National Parks & Wildlife Service, NSW, Melbourne.

Barson M. M., Randall L. A. & Bordas V. (2000) Landcover change in Australia, results of the collaborative Bureau of Rural Sciences - state agencies' project on remote sensing of land cover change. Bureau of Rural Sciences, Canberra.

Basinski J. J. (1963) Cotton growing in Australia: an agronomic survey. CSIRO, Melbourne.

Bastin G. N. & Ludwig J. A. (2006) Problems and prospects for mapping vegetation condition in Australia's arid rangelands. *Ecological Management & Restoration* **7**, S71-S4.

Batjes N. H. (1996) Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science* **47**, 151-63.

Batjes N. H. (1998) Mitigation of atmospheric CO2 concentrations by increased carbon sequestration in the soil. *Biology and Fertility of Soils* **27**, 230-5.

Beadle N. C. W. (1948) *The Vegetation and Pastures of Western New South Wales with Special Reference to Soil Erosion.* Thomas Henry Tennant, Sydney.

Beadle N. C. W. (1981) *The Vegetation of Australia*. Gustav Fischer Verlag, Stuttgart.

Bedward M. & Ellis M. (2009) Vegetation Information System (YETI) Version 3.2. (ed Department of Environment Climate Change and Water), Sydney.

Belward A. S., Taylor J. C., Stuttard M. J., Bignal E., Mathews J. & Curtis D. (1990) An unsupervised approach to the classification of semi-natural vegetation from Landsat Thematic Mapper data: A pilot study on Islay. *International Journal of Remote Sensing* **11**, 429-45.

Bennett E. M., Peterson G. D. & Gordon L. J. (2009) Understanding relationships among multiple ecosystem services *Ecology Letters* **12**, 1394-404.

Benson J. (1991) The effect of 200 years of European settlement on the vegetation and flora of New South Wales. *Cunninghamiana* **2**, 343-70.

Benson J. S. (2006) New South Wales Vegetation Classification and Assessment: Introduction - the classification, database, assessment of protected areas and threat status of plant communities. *Cunninghamia* **9**, 331-82.

Benson J. S. & Redpath P. A. (1995) The nature of pre-European native vegetation in south-eastern Australia: a critique of Ryan, D.G., Ryan, J.R. and Starr, B.J. (1995) The Australian landscape - observations of explorers and early settlers. *Cunninghamia* **5**, 285-328.

Bentley J. M. & Catterall C. P. (1997) The use of bushland, corridors, and linear remnants by birds in southeastern Queensland, Australia. *Conservation Biology* **11**, 1173-89.

Bergkamp G. (1998) A hierarchical view of the interactions of runoff and infiltration with vegetation and microtopography in semiarid shrublands. *Catena* **33**, 201-20.

Berry L. (2001) Edge effects on the distribution and abundance of birds in a southern Victorian forest *Wildlife Research* **28**, 239-45.

Bingham B. B. & Sawyer J. O. (1988) Volume and mass of decaying logs in an upland old-growth redwood forest. *Canadian Journal of Forest Research* **18**, 1649-51.

Bird A. C., Taylor J. C. & Brewer T. R. (2000) Mapping national park landscape from ground, air and space. *International Journal of Remote Sensing* **21**, 2719-36.

Bird S. B., Herrick J. E., Wander M. M. & Wright S. F. (2002) Spatial heterogeneity of aggregate stability and soil carbon in semi-arid rangeland. *Environmental Pollution* **116**, 445-55.

Birk E. M. & Simpson R. W. (1980) Steady state and the continuous input model of litter accumulation and decomposition in Australian eucalypt forests. *Ecology* **61**, 481-5.

Björklund J., Limburg K. & Rydberg T. (1999) Impact of production intensity on the ability of the agricultural landscape to generate ecosystem services: an example from Sweden. *Ecological Economics* **29**, 269-91.

Black C. A., Evans D. D., White L., Ensminger L. E. & Clark F. E. (1965) *Methods of Soil Analysis: Physical and Mineralogical Properties, Including Statistics* *of Measurement and Sampling.* American Society of Agronomy Inc., Wisconsin.

Boland B. J., Brooker M. I. H., Chippendale G. M., Hall N., Hyland B. P. M., Johnson R. D., Kleinig D. A., McDonald M. W. & Turner J. D. (2006) *Forest Trees of Australia*. CSIRO Publishing, Canberra.

Bossuyt H., Denef K., Six J., Frey S. D., Merckx R. & Paustian K. (2001) Influence of microbial populations and residue quality on aggregate stability. *Applied Soil Ecology* **16**, 195-208.

Bowen B. S. & Kruse A. D. (1993) Effects of grazing on nesting by upland sandpipers in southcentral North Dakota. *The Journal of Wildlife Management* **57**, 291-301.

Bowman D. M. J. S. (1998) Tansley Review No. 101: The impact of Aboriginal landscape burning on the Australian biota. *New Phytologist* **140**, 385-410.

Breidenrich C., Magraw D., Rowley A. & Rubin J. W. (1998) The Kyoto Protocol to the United Nations Framework Convention on Climate Change. *The Americal Journal of International Law* **92**, 315-31.

Briggs S. V. & Maher M. T. (1983) Litter fall and leaf decomposition in a River Red Gum (*Eucalyptus camaldulensis*) swamp. *Australian Journal of Botany* **31**, 307-16.

Bronick C. J. & Lal R. (2005) Soil structure and management: a review. *Geoderma* **124**, 3-22.

Brooks T. M., Mittermeier R. A., da Fonseca G. A. B., Gerlach J., Hoffmann M., Lamoreux J. F., Mittermeier C. G., Pilgrim J. D. & Rodrigues A. S. L. (2006) Global biodiversity conservation priorities. *Science* **313**, 58-61.

Brown S. (2001) Measuring carbon in forests: current status and future challenges. *Environmental Pollution* **116**, 363-72.

Burbridge A. A. & Wallace K. J. (1995) Practical methods for conserving biodiversity. In: *Conserving Biodiversity: Threats and Solutions* (eds R. A. Bradstock, T. D. Auld, D. A. Keith, R. T. Kingsford, D. Lunney and D. P. Sivertsen). Surrey Beatty & Sons Pty Limited, Chipping Norton.

Bureau of Meteorology. (2007) National Archives of Australia, Rainfall and Climate Files. Climate and Consultancy Section in the NSW Office of the Bureau of Meteorology,.

Bureau of Meteorology. (2008) National Archives of Australia, Rainfall and Climate Files. Climate and Consultancy Section in the NSW Office of the Bureau of Meteorology,.

Burgman M. A., Possingham H. P., Lynch A. J. J., Keith D. A., McCarthy M. A., Hopper S. D., Dury W. L., Passioura J. A. & Devries R. J. (2001) A method for setting the size of plant conservation target areas. *Conservation Biology* **15**, 603-16.

Burton J. R., Perrens S. J. & Fisher I. H. (1977) Proposals for a multi-objective approach to the planning of a flood mitigation scheme for the Namoi Valley. University of New England, School of Natural Resources, Armidale.

Butler C. D. & Oluoch-Kosura W. (2006) Linking future ecosystem services and future human well-being. *Ecology and Society* **11**, online.

Campbell M. C. (1989) A review of information relevant to the Mitchell grass plain rangelands of western New South Wales. In: *Rangeland Review: Mitchell Grass Plains*.

Carey E. V., Sala A., Keane R. & Callaway R. M. (2001) Are old forests underestimated as global carbon sinks? *Global Change Biology* **7**, 339-44.

Carpenter S. R., DeFries R., Dietz T., Mooney H. A., Polasky S., Reid W. V. & Scholes R. J. (2006) Millennium ecosystem assessment: research needs. *Science* **314**.

Carpenter S. R., Mooney H. A., Agard J., Capistrano D., DeFries R. S., Díaz S., Dietz T., Duraiappah A. K., Oteng-Yeboah A., Pereira H. M., Perrings C., Ried W. V., Sarukhán J., Scholes R. J. & Whyte A. (2009) Science for managing ecosystem services: beyond the millenium ecosystem assessment. *Proceedings of the National Academy of Science* **106**, 1305-12s.

Carter V. G. & Dale T. (1974) *Topsoil and Civilisation*. University of Oaklahoma Press.

Cass A. (1999) Interpretation of some soil physical indicators for assessing soil physical fertility. In: *Soil Analysis and Interpretation Manual* (eds K. I. Peverill, L. A. Sparrow and D. J. Reuter). CSIRO Publishing, Collingwood.

Cerdà A. (1998) Soil aggregate stability under different Mediterranean vegetation types. *Catena* **32**, 73-86.

Chan K. M. A., Shaw M. R., Cameron D. R., Underwood E. C. & Daily G. C. (2006) Conservation planning for ecosystem services. *PLoS Biology* **4**, 370 [online].

Chan K. Y., Bellotti W. D. & Roberts W. P. (1988) Changes in surface soil properties of vertisols under dryland cropping in a semi-arid environment. *Australian Journal of Soil Research* **26**, 509-18.

Chan K. Y., Bowman A. M. & Friend J. J. (1997) Restoration of soil fertility of degraded vertisols using a pasture including a native grass (*Astrebla lappacea*). *Tropical Grasslands* **31**, 145-55.

Chan K. Y., Hodgson A. S. & Bowman A. M. (1995) Degradation of Australian vertisols after conversion from native grassland (*Astrebla lappacea*) to continuous cropping in a semi-arid subtropical environment. *Tropical Grasslands* **29**, 210-7.

Charley J. L. & Cowling S. W. (1968) Changes in the soil nutrient status resulting from overgrazing and their consequences in plant communities of semi-arid areas. *Ecological Society of Australia* **3**, 28-38.

Charman P. E. V. & Roper M. M. (2000) *Soil Organic Matter*. Oxford University Press, New York.

Chorom M. & Rengasamy P. (1997) Carbonate chemistry, pH and physical properties of an alkaline soil as affected by varoius amendments. *Australian Journal of Soil Research* **35**, 149-61.

Cihlar J. (2000) Land cover mapping of large areas from satellites: status and research priorities. *International Journal of Remote Sensing* **21**, 1093–114.

Clark K. R. & Gourley R. N. (2006) *Primer 6 User Manual/Tutorial*. PRIMER-E, Plymouth.

Clark K. R. & Warwick R. M. (2006) *Change in Marine Communities: An Approach to Statistical Analysis and Interpretation.* PRIMER-E, Plymouth.

Clements F. E. (1916) Plant succession: an analysis of the development of vegetation. *Carnegie Institution of Washington* **242**, 1-512.

Commonwealth of Australia. (1997) Nationally agreed criteria for the establishment of a comprehensive, adequate and representative reserve system for forests in Australia. A report by the Joint ANZECC/MCFFA National Forest Policy Statement Implementation Sub-Committee, Canberra.

Commonwealth of Australia. (2009) Australia's strategy for the national reserve system 2009–2030.

Congalton R. G. (1991) A review of assessing the accuracy of classifications of remotely sensed data. *Remote Sensing of Environment* **37**, 35-46.

Cook G. D., So H. B. & Dalal R. C. (1992) Structural degradation of two vertisols under continuous cultivation. *Soil and Tillage Research* **24**, 47-64.

Costanza R., d'Arge R., de Groot R., Farber S., Grasso M., Hannon B., Limburg K., Naeem S., O'Neill R., V, Paruelo J., Raskin R. G., Sutton P. & van den Belt M. (1998) The value of ecosystem services: putting the issues in perspective. *Ecological Economics* **25**, 67-72.

Costanza R., d'Arge R., de Groot R. S., Farber S., Grasso M., Hannon B., Limburg K., Naeem S., O'Neill R. V., Paruelo J., Raskin R. G., Sutton P. & van den Belt M. (1997) The value of the world's ecosystem services and natural capital. *Nature* **387**, 253-60.

Cotton Australia. (2000) Australian cotton industry best management practices manual. Cotton Australia, Surry Hills, NSW.

Cotton Australia. (2006) The development and assessment of the cotton BMP program into a comprehensive environmental management system (EMS) through the development of a land and water module: final report. Department of Agriculture Fisheries and Forestry, Canberra.

Cotton Australia. (2009) Cotton Australia strategic plan 2009-2013 (http://www.cottonaustralia.com.au/library/publications/Cotton_Australia_Strategic_Plan_2009-2013.pdf). Cotton Australia, Mascot, NSW.

Coughlan K. J. (1984) The properties and utilization of cracking clay soils. In: *Reviews in Rural Science 5: The Properties and Utilization of Cracking Clay*

Soils (eds J. W. McGarity, E. H. Hoult and H. B. So). University of New England, Armidale.

Coughlan K. J. & Loch R. J. (1984) The relationship between aggregation and other soil properties in cracking clay soils. *Australian Journal of Soil Research* **22**, 59-69.

Coughlan K. J., McGarry D. & Smith G. D. (1987) The physical and mechanical characterisation of vertisols. In: *First Regional Seminar on Management of Vertisols under Semi-Arid Conditions*. IBSRAM, Nairobi, Kenya.

Coulombe C. E., Dixon J. B. & Wilding L. P. (1996) *Mineralogy and chemistry of vertosols*. Elsevier, Amsterdam.

Council of Australian Governments (COAG). (2000) A national action plan for salinity and water quality.

Coupland R. T. (1993) Review. In: *Ecosystems of the World: Eastern Hemisphere and Résumé* (ed R. T. Coupland) pp. 471-82. Elsevier, Amsterdam.

Cox S. J., Sivertsen D. P. & Bedward M. (2001) Clearing of native woody vegetation in the New South Wales northern wheatbelt: extent, rate of loss and implications for biodiversity conservation. *Cunninghamia* **7**, 101-55.

Crapper P. F., Beavis S. G. & Zhang L. (1999) The relationship between climate and streamflow in the Namoi basin. *Environment International* **25**, 827-39.

Crawford P. (2008) Lippia (*Phyla canescens*) management: Challanges, opportunities and strategies. Queensland Murray-Darling Committee for the National Lippia Working Group, Toowoomba, Queensland.

CRDC. (2005) Taking responsibility for our future: The Australian cotton industry action response to the second Australian cotton industry environmental audit 2003. Cotton Research and Development Corporation, Narrabri, NSW.

Croft S. J., Pedler J. A. & Milne T. I. (2009) *Bushland condition monitoring manual: Murray Darling Basin, South Australia*. The Nature Conservation Society of South Australia, Inc.

Crowley T. J. (2000) Causes of climate change over the past 1000 years. *Science* **289**, 270-7.

Cunningham G. (1976) Some concepts for range assessment in Australia. *The Rangeland Journal* **1**, 60-3.

Cunningham G. M., Mulham W. E., Milthorpe P. L. & Leigh J. H. (1992) *Plants of Western New South Wales*. Inkata Press, Melbourne.

Curtin D. H., Steppuhn H. & Selles F. (1994) Effects of magnesium on cation selectivity and structural stability of sodic soils. *Soil Science Society of America Proceedings* **58**, 730-7.

Daily G. (1997) *Nature's Services: Societal Dependence on Natural Ecosystems*. Island Press, Washington.

Daily G. (2000a) Management objectives for the protection of ecosystem services. *Environmental Science and Policy* **3**, 333-9.

Daily G. C. (1999) Developing a scientific basis for managing Earth's life support systems. *Conservation Ecology* **3**, 14. [online].

Daily G. C. (2000b) Management objectives for the protection of ecosystem services. *Environmental Science & Policy* **3**, 333-9.

Dalal R. C. (1989) Long-term effects of no-tillage, crop residue and nitrogen application on properties of a vertosol. *Soil Science Society of America Proceedings* **53**, 1511-5.

Dalal R. C. & Bridge B. J. (1996) *Aggregation and organic matter storage in sub-humid and semi-arid soils*. CRC Press Inc., New York.

Dalal R. C. & Chan K. Y. (2001) Soil organic matter in rainfed cropping systems of the Australian cereal belt. *Australian Journal of Soil Research* **39**, 435-64.

Dambach C. A. (1944) A ten-year ecological study of adjoining grazed and ungrazed woodlands in northeastern Ohio. *Ecological Monographs* **14**, 256-70.

Dambach C. A. & Good E. E. (1940) The effect of certain land use practices on populations of breeding birds in southwestern Ohio. *The Journal of Wildlife Management* **4**, 63-76.

Davidson I., Scammell A., O'Shannassy P., Mullins M. & Learmouth S. (2005) Travelling stock reserves: refuges for stock and biodiversity? *Ecological Management and Restoration* **6**, 5-15.

Day P. R. (1965) Particle fractionation and particle-size analysis. In: *Methods of Soil Analysis, Part 1: Physical and Mineralogical Properties, Including Statistics of Measurement and Sampling.* (ed M. A. Black) pp. 545-67. American Society of Agronomy, Wisconsin.

de Groot R. S., Wilson M. A. & Boumans R. M. J. (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* **41**, 393-408.

DeFries R. S., Foley J. A. & Asner G. P. (2004) Balancing human needs and ecosystem function. *Frontiers in Ecology and the Environment* **2**, 249-57.

Delaney M., Brown S., Lugo A. E., Torres-Lezama A. & Quintero N. B. (1998) The quantity and turnover of dead wood in permanent forest plots in six life zones of Venezuela. *Biotropica* **30**, 2-11.

Department of Natural Resources. (2006) Great Artesian Basin sustainability initiative: Cap and pipe the bores.

DEWHA. (2010) Preliminary EPBC Act assessment of the Coolibah-Black box woodlands of the Darling Riverine Plains and the Brigalow Belt South bioregions. Department of the Environment, Water, Heritage and the Arts, Canberra.

Diamond J. (2005) *Collapse, How Societies Choose to Fail*. McPherson's Printing Group, Maryborough.

DLWC. (1999) North west vegetation mapping - Walgett Shire east. Department of Land and Water Conservation, Gunnedah.

Dorrough J. & Moxham C. (2005) Eucalypt establishment in agricultural landscapes and implications for landscape-scale restoration. *Biological Conservation* **123**, 55-66.

Dow D. D. (1979) Agonisitc and spacing behaviour of the noisy miner, *Manorina melanocephala*, a communally breeding honeyeater. *The Ibis* **121**, 423-36.

Doyle R. B. & Habracken F. M. (1993) The distribution of sodic soils in Tasmania. *Australian Journal of Soil Research* **31**, 931-47.

Driese K. L., Reiners W. A., Merrill E. H. & Gerow K. G. (1997) A digital land cover map of Wyoming, USA: A tool for vegetation analysis. *Journal of Vegetation Science* **8**, 133-46.

DSE. (2004) Vegetation quality assessment manual - guidelines for applying the Habitat Hectares scoring method. Version 1.3. Victorian Government, Department of Sustainability and Environment, Melbourne.

DSE. (2006) River red gum forests factsheet. Victorian Department of Sustainability and Environment, Melbourne, Victoria.

Dudley N., Schlaepfer R., Jackson W., Jeanrenaud J. & Stolton S. (2006) *Forest quality: assessing forests at a landscape scale*. Earthscan, London.

Eamus D., Macinnis-Ng C. M. O., Hose G. C., Zeppel M. J. B., Taylor D. T. & Murray B. R. (2005) TURNER REVIEW No. 9 Ecosystem services: an ecophysiological examination. *Australian Journal of Botany* **53**, 1-19.

Earl J. (2003) The distribution and impacts of Lippia (*Phyla canescens*) in the Murray Darling System. Agricultural Information & Monitoring Services for the Lippia Working Group.

Eco Logical Australia. (2006) Vegetation mapping and survey data audit and gap analysis for the Namoi and Border Rivers-Gwydir CMAs. Eco Logical Australia, Sutherland, NSW.

Eco Logical Australia. (2009) Riverine vegetation in the Namoi catchment: An assessment of type and condition. Eco Logical Australia, Coffs Harbour, New South Wales.

Egoh B., Reyers B., Rouget M., Bode M. & Richardson D. M. (2009) Spatial congruence between biodiversity and ecosystem services in South Africa. *Biological Conservation* **142**, 553-62.

Egoh B., Rouget M., Reyers B., Knight A. T., Cowling R. M., van Jaarsveld A. S. & Welz A. (2007) Integrating ecosystem services into conservation assessments: a review. *Ecological Economics* **63**, 714-21.

Ehrlich P. R. & Ehrlich A. H. (1981) *Extinction*. Random House, New York.

El Serafy S. (1998) Pricing the invaluable: The value of the world's ecosystem servies and natural capital. *Ecological Economics* **25**, 25-7.

Eldridge D. J. & Koen T. B. (1993) Run-off and sediment yield from a semi-arid woodland in eastern Australia. II. Variation in some soil hydrological properties along a gradient in soil surface condition. *The Rangeland Journal* **15**, 234-46.

Emerson W. W. (1967) A classification of soil aggregates based on their coherence in water. *Australian Journal of Soil Research* **5**, 47-57.

Emerson W. W. (1983) *Inter-particle bonding*. Academic Press, London.

Eviner V. T. & Chapin F. S. I. (2002) The influence of plant species, fertilization and elevated CO_2 on soil aggregate stability. *Plant and Soil* **246**, 211-9.

Eyre T. J., Kelly A. I. & Nelder V. J. (2008) BioCondition: A terrestrial vegetation condition assessment tool for biodiversity in Queensland, assessment manual version 1.6. Environmental Protection Agency, Biodiversity Sciences Unit, Brisbane.

Facelli J. M. & Brock D. J. (2000) Patch dynamics in arid lands: localized effects of *Acacia papyrocarpa* on soils and vegetation of open woodlands of South Australia. *Ecography* **23**, 479-91.

Fanning P. (1994) Long-term contempory erosion rates in an rangelands environment in western New South Wales, Australia. *Journal of Arid Environments* **28**, 173-87.

Farber S., Costanza R., Childers D. L., Erickson J., Gross K., Grove M., Hopkinson C. S., Kahn J., Pincetl S., Troy A., Warren P. & Wilson M. (2006) Linking ecology and economics for ecosystem management. *BioScience* **56**, 121-33.

Farber S. C., Costanza R. & Wilson M. A. (2002) Economic and ecological concepts for valuing ecosystem services. *Ecological Economics* **41**, 375-92.

Ferry J. (1949) *European Expansion and Settlement, 1821-1850: A History of the North West Plains of New South Wales.* Koala Instant Printing, Sydney.

Ferry J. (1978) *Walgett Before the Motor Car*. Pictorial Publishers, Sydney.

Field D. J. (2000) The Bricks and Mortar of Vertosols: The Characteristics of Aggregation and Assessing Aggregate Stability of Vertosols used for Cotton Production. University of Sydney, Sydney.

Field D. J., McKenzie D. C. & Koppi A. J. (1997) Development of an improved vertosol stability test for SOILpak. *Australian Journal of Soil Research* **35**, 843-52.

Fischer A. M. & Goldney D. C. (1997) Use by birds of riparian vegetation in an extensively fragmented landscape. *Pacific Conservation Biology* **3**, 275-88.

Fischer J. & Lindenmayer D. B. (2002) Small patches can be valuable for biodiversity conservation: two case studies on birds in southeastern Australia. *Biological Conservation* **106**, 129-36.

Fischer J. & Lindenmayer D. B. (2007) Landscape modification and habitat fragmentation: A synthesis. *Global Ecology and Biogeography* **16** 265-80.

Fischer J., Lindenmayer D. B. & Manning A. D. (2006) Biodiversity, ecosystem function, and resilience: ten guiding principles for commodity production landscapes. *Frontiers in Ecology and the Environment* **4**, 80-6.

Fischlin A., Midgley G. F., Price J. T., Leemans R., Copal B., Turley C., Rounsevell M. D. A., Duke O. P., Tarazona J. & Velichko A. A. (2007) Ecosystems, their properties, goods and services. In: *Climate change 2007: Impacts, adaption and vulnerability. Contribution of the Working Group II to the 4th Assessment Report of the Intergovernmental Panel of Climate Change (IPCC)* (eds M. L. Parry, O. F. Canziani, I. P. Palutikof, P. J. van der Linden and C. E. Hanson) pp. 211-72. Cambridge University Press, Cambridge, UK.

Foley J. A., DeFries R., Asner G. P., Barford C., Bonan G., Carpenter S. R., Chapin F. S., Coe M. T., Daily G. C., Gibbs H. K., Helkowski J. H., Holloway T., Howard E. A., Kucharik C. J., Monfreda C., Patz J. A., Prentice I. C., Ramankutty N. & Snyder P. K. (2005) Global consequences of land use. *Science* **309**, 570-4.

Fondell T. F. & Ball I. J. (2004) Density and success of bird nests relative to grazing on western Montana grasslands. *Biological Conservation* **117**, 203-13.

Food and Agriculture Organisation. (2002) World agriculture: Towards 2015/2030: Summary report. Food and Agriculture Organisation, Rome.

Ford H. A., Barrett G. W., Saunders D. & Recher H. F. (2001) Why have birds in the woodlands of southern Australia declined? *Biological Conservation* **97**, 71-88.

Fox T. R. (2000) Sustained productivity in intensively managed forest plantations. *Forest Ecology and Management* **138**, 187-202.

Freebairn D. M., Loch R. J. & Silburn D. M. (1996) *Soil erosion and soil conservation for vertosols*. Elsevier, Amsterdam.

Freebairn D. M. & Wockner G. H. (1986) A study of soil erosion on vertisols of the eastern Darling Downs, Queensland. I Effects of surface conditions on soil movement within contour bay catchments. *Australian Journal of Soil Research* **24**, 135-58.

Freudenberger D. (1998) Scoping the management and research needs of the coolibah woodlands in the Murray-Darling Basin. Murray-Darling Basin Commission, Lyneham.

Freudenberger D. & Drew A. (2001) Bird surveys in travelling stock routes and reserves on the northwest slopes and plains: perspectives on the conservation value TSRs. CSIRO for the Northern Slopes Rural Lands Protection Boards, NSW.

Geeves G. W., Craze B. & Hamilton G. J. (2000) Soil physical properties. In: *Soils: Their Properties and Management* (eds P. E. V. Charman and B. Murphy). Oxford University Press, Melbourne.

GHD. (2003) Second Australian cotton industry environmental audit. Cotton Research and Development Corporation, Chatswood.

Gibb Environmental Sciences & Arbour International. (1991) An environmental audit of the Australian cotton industry, report prepared for the Australian Cotton Foundation.

Gibbons P., Ayres D., Sneddon J., Doyle S. & Briggs S. (2008a) BioMetric: A terrestrial biodiversity assessment tool for the NSW native vegetation assessment tool, version 2.0, Operational Manual. NSW Department of Environment and Climate Change, Canberra.

Gibbons P. & Freudenberger D. (2006) An overview of methods used to assess vegetation condition at the scale of the site. *Ecological Management and Restoration* **7**, S10-S7.

Gibbons P., Lindenmayer D. B., Fischer J., Manning A. D., Weinberg A., Seddon J., Ryan P. & Barrett G. (2008b) The future of scattered trees in agricultural landscapes. *Conservation Biology* **22**, 1309-19.

Gibbons P., Zerger A., Jones S. & Ryan P. (2006) Mapping vegetation condition in the context of biodiversity conservation. *Ecological Management and Restoration* **7**, S1-S2.

Gilbee A. & Goodson P. (1992) Mapping tree cover across Victoria using thematic mapper digital data and a geographic information system. In: *Proceedings of the 6th Australasian Remote Sensing Conference*. Conference Committee, Wellington, New Zealand.

Gillespie R. (2000) Economic values of native vegetation, background paper number 4. Native Vegetation Advisory Council, Sydney.

Glanznig A. (1995) Native vegetation clearance, habitat loss and biodiversity decline, an overview of recent vegetation clearance in Australia and its implications for biodiversity. Paper No. 6. Biodiversity Unit, Sydney.

Good E. E. & Dambach C. A. (1943) Effect of land use practices on breeding bird populations in Ohio. *The Journal of Wildlife Management* **7**, 291-7.

Gourley C. J. P. (1999) Potassium. In: *Soil Analysis an Interpretation Manual* (eds K. I. Peverill, L. A. Sparrow and D. J. Reuter). CSIRO Publishing, Collingwood.

Gracía-Llorente M., Martín-López B., González J. A. & Alcorlo P. (2008) Social perceptions of the impacts and benefits of invasive alien species: Implications for management. *Biological Conservation* **141**, 2969-83.

Graetz R. D. & Tongway D. J. (1986) Influence of grazing management on vegetation, soil structure, nutrient distribution and the infiltration of applied rainfall in a semi-arid chenopod shrubland. *Australian Journal of Ecology* **11**, 347-60.

Grant A. R. (2005) *Tracking changes: Darling Riverine Plains bioregion New South Wales 1992-2002*. Australian Collaborative Rangeland Information System, Dubbo.

Green E. P., Mumby P. J., Edwards A. J. & Clark C. D. (1996) A review of remote sensing for the assessment and management of tropical coastal resources. *Coastal Management* **24**, 1-40.

Greene R. S. B. & Tongway D. J. (1989) The significance of (surface) physical and chemical properties in determining soil surface condition of red earths in rangelands. *Australian Journal of Soil Research* **27**, 213-25.

Greenwood K. L. & McKenzie B. M. (2001) Grazing effects on soil physical properties and the consequences for pastures: A review. *Australian Journal of Experimental Agriculture* **41**, 1231-50.

Greiner R., Gordon I. & Cocklin C. (2009) Ecosystem services from tropical savannas: economic opportunities through payments for environmental services. *The Rangeland Journal* **31**, 51-9.

Grey M. J., Clarke M. F. & Loyn R. H. (1997) Initial changes in the avian communities of remnant eucalypt woodlands following a feduction in the abundance of noisy miners, *Manorina melanocephala*. *Wildlife Research* **24**, 631-48.

Hairsine P. (2001) The prevention of soil erosion as an ecosystem service. In: *Natural Assets: An Inventory of Ecosystem Goods and Services in the Goulburn Broken Catchment* (eds C. Binning, S. Cork, R. Parry and D. Shelton). CSIRO, Canberra.

Harden G. J. (1993) *Flora of New South Wales, Volume 4*. University of New South Wales, Sydney.

Harden G. J. (2000) *Flora of New South Wales, Volume 1*. University of New South Wales, Sydney.

Harden G. J. (2002a) *Flora of New South Wales, Volume 2*. University of Newcastle, Sydney.

Harden G. J. (2002b) *Flora of New South Wales, Volume 3*. University of New South Wales, Sydney.

Harmon M., Ferrel W. K. & Franklin J. F. (1990) Effects on carbon storage of conversion of old-growth forests to young forests. *Science* **247**, 699-702.

Harmon M. E., Franklin J. F., Swanson F. J., Sollins P., Gregory S. V., Lattin J. D., Anderson N. M., Cline S. P., Aumen N. G., Sedell J. R., Lienkaemper G. W., Cromack J., K. & Cummins K. W. (1986) *Ecoloogy of Coarse Woody Debris in Temperate Ecosystems*. Academic Press Inc. Ltd, London.

Harrington G. N., Mills D. M. D., Pressland A. J. & Hodgkinson K. C. (1984) Semi-arid woodlands. In: *Management of Australia's Rangelands* (eds G. N. Harrington, A. D. Wilson and M. D. Young). CSIRO Publishing, Melbourne.

Harrington G. N., Oxley R. E. & Tongway D. J. (1979) The effects of European settlement and domestic livestock on the biological system in poplar box (*Eucalyptus populnea*) lands. *Australian Rangeland Journal* **1**, 271-9.

Hassink J., Whitmore A. P. & Kubát J. (1997) Size and density fractionation of soil organic matter and the physical capacity of soils to protect organic matter. *European Journal of Agronomy* **7**, 189-99.

Heal G. (2000) Valuing ecosystem services. *Ecosystems* **3**, 24-30.

Heal G., Daily G. C., Ehrlich P. R., Salzman J., Boggs C., Hellmann J., Hughes J., Kremen C. & Ricketts T. (2001) Protecting natural capital through ecosystem services districts. *Stanford Environmental Law Journal* **20**, 333-64.

Hector A., Schmid B., Beierkuhnlein C., Caldeira M. C., Diemer M., Dimitrakopoulos P. G., Finn J. A., Freitas H., Giller P. S., Good J., Harris R., Hogberg P., Huss-Danell K., Joshi J., Jumpponen A., rner C., Leadley P. W., Loreau M., Minns A., Mulder C. P., O'Donovan G., Otway S. J., Pereira J. S., Prinz A., Read D. J., Scherer-Lorenzen M., Schulze E.-D., Siamantziouras A.-S. D., Spehn E. M., Terry A. C., Troumbis A. Y., Woodward F. I., Yachi S. & Lawton J. (1999) Plant diversity and productivity experiments in European grasslands. *Science* **286**, 1123-7.

Helling C. S., Chesters G. & Corey R. B. (1964) Contributions of organic matter and clay to soil cation exchange capacity as affected by the pH of the saturating solution. *Soil Science Society of America Journal* **28**, 517-20.

Henle K., Davies K. F., Kleyer M., Margules C. & Settele J. (2004) Predictors of species sensitivity to fragmentation. *Biodiversity and Conservation* **13**, 207-51.

Heywood V. H. (1995) *The Global Biodiversity Assessment. United Nations Environment Programme*. Cambridge University Press, Cambridge.

Higgins P. J. & Peter J. M. (2002) *Handbook of Australian, New Zealand and Antarctic Birds. Volume 6: Pardalotes to shrike-thrushes*. Oxford University Press, Melbourne.

Higgins P. J., Peter J. M. & Cowling S. J. (2006) *Handbook of Australian, New Zealand and Antarctic Birds. Volume 7: Boatbills to Starlings*. Oxford University Press, Melbourne.

Hilderbrand R. H., Watts A. C. & Randle A. M. (2005) The myths of restoration ecology. *Ecology and Society* **10**, 19-29.

Hill S. J., Tung P. J. & Leishman M. R. (2005) Relationships between anthropogenic disturbance, soil properties and plant invasion in endangered Cumberland Plain woodland, Australia. *Austral Ecology* **30**, 775-88.

Hobbs R. J. (2001) Synergisms among habitat fragmentation, livestock grazing, and biotic invasions in southwestern Australia. *Conservation Biology* **15**, 1522-8.

Hobbs R. J. & Hopkins A. J. M. (1988) From frountier to fragments: European impact on Australia's vegetation. In: *Australian Ecosystems: 200 Years of Utilisation, Degradation and Reconstruction* (eds D. A. Saunders, A. J. M. Hopkins and R. A. How) pp. 93-114. Ecological Society of Australia, Geraldton, Western Australia.

Hobbs R. J. & Kristjanson L. J. (2003) Triage: how do we prioritize health care for landscapes? *Ecological Management & Restoration* **4** S39-45.

Hobbs R. J. & Yates C. J. (2003) TURNER REVIEW No. 7. Impacts of ecosystem fragmentation on plant populations: generalising the idiosyncratic. *Australian Journal of Botany* **51**, 471-88.

Homer C. G., Ramsey R. D., Edwards T. C. & Falconer A. (1997) Landscape cover-type modelling using a multi-scene Thematic Mapper mosaic. *Photogrammetric Engineering and Remote Sensing* **63**, 59-67.

Hooper B. P. & Duggin J. A. (1996) Ecological riverine floodplain zoning. *Land Use Policy* **13**, 87-99.

Hopkins A. (1999) National Land and Water Resources Audit Vegetation Theme. Condition of vegetation: a discussion paper. Department of Conservation and Land Management, Wanneroo.

Houghton R. A. (1999) The annual net flux of carbon to the atmosphere from changes in land use. *Tellus* **51B**, 298-313.

House J. I., Prentice I. C. & Quéré C. L. (2002) Maximum impacts of future reforestation or deforestation on atmospheric CO_2. *Global Change Biology* **8**, 1047-52.

Howe R. W. (1984) Local dynamics of bird assemblages in small forest habitat islands in Australia and North America. *Ecology* **65**, 1585-601.

Hubble G. D. (1984) The cracking clay soils: Definition, distribution, nature, genesis and use. In: *Reviews in Rural Science 5: The Properties and Utilization of Cracking Clay Soils* (eds J. W. McGarity, E. H. Hoult and H. B. So). University of New England, Armidale.

Hunt G. L. (1980) *When Narrabri Was Young:A Story of Narrabri from 1818 to 1900*. Narrabri Municipal Council, Narrabri.

Huston M. (1979) A general hypothesis of species diversity. *The American Naturalist* **113**, 81-101.

Image S. (2003) Processing levels and location accuracy, Technical Sheet.

International Society of Soil Science. (1929) Minutes of the First Commission Meetings. In: *Proceedings of the International Society of Soil Science. Volume 4* pp. 215-20. International Congress of Soil Science, Washington DC.

IPCC. (2007) Climate change 2007: Fouth assessment synthesis report. (ed International Panel on Climate Change).

Isbell R. F. (1996) *The Australian Soil Classification Revised Edition*. CSIRO Publishing, Collingwood.

Jackson J. & Ash A. J. (1998) Tree-grass relationships in open eucalypt woodlands of northeastern Australia: Influence of trees on pasture productivity, forage quality and species distribution. *Agroforestry Systems* **40**, 159-76.

Jackson L., Rosenstock T., Thomas M., Wright J. & Symstad A. J. (2009) Managed ecosystems: biodiversity and ecosystem functions in landscapes modified by human use. In: *Biodiversity, Ecosystem Functioning, and Human Wellbeing* (eds S. Naeem, D. E. Bunker, A. Hector, M. Loreau and C. Perrings). Oxford University Press, New York.

Jacobs S. W. L., Whalley R. D. B. & Wheeler D. J. B. (2008) *Grasses of New South Wales*. University of New England, Armidale.

Jansen A. & Robertson A. I. (2001) Riparian bird communities in relation to land management practices in floodplain woodlands of south-eastern Australia. *Biological Conservation* **100**, 173-85.

Jastrow J. D. & Miller R. M. (1998) Soil aggregate stabilization and carbon sequestration: Feedbacks through organominaral associations. In: *Soil Processes and the Carbon Cycle* (eds R. Lal, J. M. Kimble, R. F. Follett and B. A. Stewart). CRC Press, London.

Jensen J. R. (2005) *Introductory Digital Image Processing: A Remote Sensing Perspective*. Pearson Prentice Hall, Upper Saddle River.

Jobbágy E. G. & Jackson R. B. (2000) The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications* **10**, 423-36.

Johansson M. E., Nilsson C. & Elisabet N. (1996) Do rivers function as corridors for plant dispersal? *Journal of Vegetation Science* **7**, 593-8.

Junor R. S., Marston D. & Donaldson S. G. (1979) *A Situation Statement of Soil Erosion in the Lower Namoi Area*. Soil Conservation Service of NSW.

Kaeiva P., Watts S., McDonald R. & Boucher T. (2007) Domesticated nature: shaping landscapes and ecosystems for human welfare. *Science* **316**, 1866-9.

Kass T. (2003) A Thematic History of Walgett Shire. Walgett Shire Council, Walgett.

Kearle A., Gosper C., Achurch H. & Laity T. (2002) Darling Riverine Plains background report: Darling Riverine Plains bioregion. In: *NSW Biodiversity Strategy*. NSW National Parks and Wildlife Service, Dubbo.

Keith D. (2004) *Ocean Shores to Desert Dunes: The Native Vegetation of New South Wales and the ACT.* Department of Environment and Conservation, Hurtsville.

Keith H., Barrett D. & Keenan R. (2000) Review of allometric relationships for estimating woody biomass for New South Wales, the Australian Capital Territory, Victoria, Tasmania and South Australia. Canberra.

Kemper W. D. & Rosenau R. C. (1986) *Aggregate stability and size distribution*. Soil Science Society of America.

Kent H. R. & William M. A. (2009) Payment for ecosystem services and the challenge of saving nature. *Conservation Biology* **23**, 785-7.

King K. L. & Hutchinson K. J. (1976) The effects of sheep stocking intensity on the abundance and distribution of mesofauna in pastures. *Journal of Applied Ecology* **13**, 41-55.

Kirkpatrick J. B. (1983) An iterative method for establishing priorites for the selection of nature reserves:An example from Tasmania. *Biological Conservation* **25**, 127-34.

Knight A. T., Smith R. J., Cowling R., M, Desmet P. G., Faith D. P., Ferrier S., Gelderblom C. M., Grantham H., Lombard A. T., Maze K., Nel J. L., Parrish

J. D., Pence G. Q. K., Possingham H., Reyers B., Rouget M., Roux D. & Wilson K. A. (2007) Improving the key biodiversity areas approach for effective conservation planning *BioScience* **57**, 256-61.

Kremen C. (2005) Managing ecosystem services: what do we need to know about their ecology? *Ecology Letters* **8**, 468-79.

Kremen C., Williams N. M., Aizen M. A., Gemmill-Herren B., LeBuhn G., Minckley R., Packer L., Potts S. G., Roulston T., Steffan-Dewenter I., Vazquez D. P., Winfree R., Adams L., Crone E. E., Greenleaf S. S., Keitt T. H., Klein A. M., Regetz J. & Ricketts T. H. (2007) Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change. *Ecology Letters* **10**, 299-314.

Kremen C., Williams N. M., Bugg R. L., Fay J. P. & Thorp R. W. (2004) The area requirements of an ecosystem service: crop pollination by native bee communities in California. *Ecology Letters* **7**, 1109-19.

Kuhnell C. A., Goulevitch B. M., Danaher T. J. & Harris D. P. (1998) Mapping woody vegetation cover over the state of Queensland using Landsat TM Imagery. In: *The 9th Australasian Remote Sensing and Photogrammetry Conference*. Causal Productions Pty Ltd, Sydney.

Lal R. (1990) Soil erosion and land degradation: the global risks. In: *Soil Degradation: Advances in Soil Science. Volume 11* (eds R. Lal and B. A. Stewart) pp. 129-72.

Lal R. (2004) Soil carbon sequestration impacts on global climate change and food security. *Science* **304**, 1623-7.

Lal R. & Stewart B. A. (1990) Soil degradation: A global threat. In: *Soil Degradation, Advances in Soil Science. Volume 11* (eds R. Lal and B. A. Stewart) pp. 13-7.

Lambeck R. J. & Saunders D. A. (1993) The role of patchiness in reconstructed wheatbelt landscapes. In: *Nature Conservation 3: Reconstruction of Fragmented Ecosystems* (eds D. Saunders, R. Hobbs and P. Ehrlich) pp. 153-61. Surrey Beatty & Sons, Sydney.

Landsberg J., James C. D., Morton S. R., Müller W. J. & Stol J. (2003) Abundance and composition of plant species along grazing gradients in Australian rangelands. *Journal of Applied Ecology* **40**, 1008-24.

Lang R. D. (1984) Runoff and soil loss from a cracking clay soil. In: *Reviews in Rural Science 5: The Properties and Utilization of Cracking Clay Soils* (eds J. W. McGarity, E. H. Hoult and H. B. So) pp. 24-8. University of New England, Armidale.

Lang R. D. (2008) Defining the original extent and floristic composition of the naturally-treeless grasslands of the Liverpool Plains, North Western Slopes, New South Wales. *Cunninghamia* **10**, 407-21.

Lark R. M. (1995) A reappraisal of unsupervised classification, II: optimal adjustment of the map legend and a neighbourhood approach for mapping legend units. *International Journal of Remote Sensing* **16**, 1445–60.

Lavelle P. & Spain A. V. (2001) *Soil Ecology*. Kluwer Academic Publishers, Dordrecht.

Law B. S. & Dickman C. R. (1998) The use of habitat mosiacs by terrestrial vertebrate fauna: implications for conservation and management. *Biodiversity and Conservation* **7**, 323-33.

Lawton J. (1994) What do species do in ecosystems? *Oikos* **71**, 367-74.

Le Bissonnais Y. & Arrouays D. (1997) Aggregate stability and assessment of soil crustability and erodibility: II. Application to humic loamy soils with various organic carbon contents. *European Journal of Soil Science* **48**, 39-48.

Leps J. & Smilauer P. (2003) *Multivariate Analysis of Ecological Data Using Canoco*. Canbridge University Press, Cambridge.

Lerdau M. & Slobodkin L. (2002) Trace gas emissions and species-dependant ecosystem services. *Trends in Ecology & Evolution* **17**, 309-12.

Lewis T., Clarke P. J., Whalley R. D. B. & Reid N. (2009) What drives plant biodiversity in the clay floodplain grasslands of NSW. *The Rangeland Journal* **31**, 329-51.

Little I. P., Ringrose-Voase A. J. & Ward W. T. (1992) Surface structure in grey clays of northwestern New South Wales in relation to micromorphology, cation suite and particle size attributes. *Australian Journal of Soil Research* **30**, 1-16.

Lloyd J., Bird M. I., Vellen L., Miranda A. C., Veenendaal E. M., Djagbletey G., Miranda H. S., Cook G. & Farquhar G. D. (2008) Contributions of woody and herbaceous vegetation to tropical savanna ecosystem productivity: a quasi-global estimate. *Tree Physiol* **28**, 451-68.

Loch R. J. (1989) Aggregate Breakdown Under Rain: Its Measurement and Interpretation. University of New England, Armidale.

Loch R. J. (1994) Effects of fallow management and cropping history on aggregate breakdown under rainfall for a range of Queensland soils. *Australian Journal of Soil Research* **32**, 1125-39.

Loveland P. & Webb J. (2003) Is there a critical level of organic matter in the agricultural soils of temperate regions: a review. *Soil and Tillage Research* **70**, 1-18.

Loyn R. H. (1987) Effects of patch area and habitat on bird abundances, species numbers and tree health in fragmented Victorian forests. In: *Nature Conservation: The Role of Remnants of Native Vegetation* (eds D. A. Saunders, G. W. Arnold, A. A. Burbridge and A. J. M. Hopkins). Surrey Beatty & Sons Pty Limited, Chipping Norton.

Luck G. W., Daily G. C. & Ehrlich P. R. (2003) Population diversity and ecosystem services. *Trends in Ecology and Evolution* **18**, 331-6.

Luck G. W., Harrington R., Harrison P. A., Kremen C., Berry P. M., Bugter R., Dawson T. P., de Bello F., DÃaz S., Feld C. K., Haslett J. R., Hering D., Kontogianni A., Lavorel S., Rounsevell M., Samways M. J., Sandin L., Settele J., Sykes M. T., van den Hove S., Vandewalle M. & Zobel M. (2009) Quantifying the contribution of organisms to the provision of ecosystem services. *BioScience* **59**, 223-35.

Ludwig J., Tongway D., Freudenberger D., Noble J. & Hodgkinson K. (1997) *Landscape Ecology, Function and Management: Principles from Australia's rangelands*

CSIRO Publishing, Canberra.

Ludwig J. A., Tongway D. J., Bastin G. N. & James C. D. (2004) Monitoring ecological indicators of rangeland functional integrity and their relation to biodiversity at local to regional scales. *Austral Ecology* **29**, 108-20.

Ludwig J. A., Tongway D. J. & Marsden S. G. (1994) A flow-filter model for simulating the conservation of limited resources in spatially heterogeneous, semi-arid landscapes. *Pacific Conservation Biology* **1**, 209-13.

Lunt I. (1991) Management of remnant lowland grasslands and grassy woodlands for nature conservation: A review. *Victorian Naturalist* **108**, 57-66.

Lunt I. D. (1997) Effects of long-term vegetation management on remnant grassy forests and anthropogenic native grasslands in south-eastern Australia. *Biological Conservation* **81**, 287-97.

Lunt I. D., Eldridge D. J., Morgan J. W. & Witt G. B. (2007) TURNER REVIEW No. 13 A framework to predict the effects of livestock grazing and grazing exclusion on conservation values in natural ecosystems in Australia. *Australian Journal of Botany* **55**, 401-15.

Lunt I. D. & Morgan J. W. (1999) Vegetation changes after 10 years of grazing exclusion and intermittent burning in a *Themeda triandra* (Poaceae) grassland reserve in south-eastern Australia. *Australian Journal of Botany* **47**, 537-52.

Luyssaert S., Schulze E. D., Borner A., Knohl A., Hessenmoller D., Law B. E., Ciais P. & Grace J. (2008) Old-growth forests as global carbon sinks. *Nature* **455**, 213-5.

MA (Millennium Ecosystem Assessment). (2003) *Ecosystems and Human Well-Being*. Island Press, Washington D.C.

MA (Millennium Ecosystem Assessment). (2005) *Ecosystems and Human Well-being. Synthesis*. Island Press, Washington, DC.

Maass J. M., Balvanera P., Castillo A., Daily G., Mooney H. A., Ehrlich P., Quesada M., Miranda A., Jaramillo V. J., García-Oliva F., Martinez-Yrizar A., Cotler H., López-Blanco J., Pérez-Jiménez A., Búrquez A., Tinoco C., Cebellos G., Barraza L., Ayala R. & Sarukhán J. (2005) Ecosystem services of tropical dry forests: Insights from long-term ecological and social research on the Pacific coast of Mexico. *Ecology and Society* **10**, online.

Mac Nally R., Soderquist T. R. & Tzaros C. (2000) The conservation value of mesic gullies in dry forest landscapes: avian assemblages in the box-ironbark ecosystem of southern Australia. *Biological Conservation* **93**, 293-302.

MacArthur R. H. & MacArthur J. W. (1961) On bird species diversity. *Ecology* **42**, 594-8.

Mack M. C. & D'Antonio C. M. (1998) Impacts of biological invasions on disturbance regimes. *Trends in Ecology and Evolution* **13**, 195-8.

Mackensen J., Bauhus J. & Webber E. (2003) Decomposition rates of coarse woody debris - A review with particular emphasis on Australian tree species. *Australian Journal of Botany* **51**, 27-37.

Maher M. T. (1995) A thin line: Should densities of coolibah and black box be controlled in the western division of New South Wales?. Report to the Western Lands Commission, Dubbo. Department of Conservation and Land Management.

Maiden J. H. (1889) *A Manual of Grasses of New South Wales*. Government Printers, Sydney.

Margules C. R. (1981) Assessment of Wildlife Conservation Values. University of York.

Margules C. R. (1989) Introduction to some Australian developments in conservation evaluation. *Biological Conservation* **50**, 1-11.

Margules C. R. & Pressey R. L. (2000) Systematic conservation planning. *Nature* **405**, 243-53s.

Marsh G. P. (1864) *Man and Nature*. Charles Scribner, New York.

Martin T. G., McIntyre S., Catterall C. P. & Possingham H. (2006) Is landscape context important for riparian conservation? Birds in grassy woodland. *Biological Conservation* **127**, 201-14.

Martin T. G. & Possingham H. P. (2005) Predicting the impact of livestock grazing on birds using foraging height data. *Journal of Applied Ecology* **42**, 400-8.

Mawhinney M. (2004) Priority environmental weeds in the Namoi catchment. Namoi Catchment Management Authority, Tamworth.

Mawhinney W. A. (2003) Restoring biodiversity in the Gwydir Wetlands through environmental flows. *Water Science and Technology* **48**, 73-81.

McCauley D. (2006) Selling out on nature. *Nature* **443**, 27-8.

McCosker R. O. (1994) Lippia (*Phyla nodiflora)*: An invasive plant of floodplain ecosystems in the Murray-Darling Basin: A report on the distribution and ecology of lippia in the lower Gwydir Valley and the Murray-Darling Basin. University of New England, Armidale.

McGarity J. W. (1950) The Nature and Distribution of Soils of the North West Namoi Region in Relation to the Factors Determining their Genesis and Future Utilisation. University of Sydney, Sydney.

McGee G. G., Leopold D. J. & Nyland R. D. (1999) Structural characteristics of old-growth, maturing, and partially cut northern hardwood forests. *Ecological Applications* **9**, 1316-29.

McHugh S. (1996) *Cottoning On: Stories of Australian Cotton Growing*. Hale & Iremonger, Sydney.

McIntyre S. & Hobbs R. (1999) A framework for conceptualizing human effects on landscapes and its relevance to management and research models. *Conservation Biology* **13**, 1282-92.

McIntyre S. & Hoobs R. (2000) Human impacts on landscapes: Matrix condition and management priorities. In: *Conservation in Production Environments: Managing the Matrix* (eds J. L. Craig, N. Mitchell and D. A. Saunders). Surrey Beatty & Sons Pty Limited, Chipping Norton.

McIntyre S. & Lavorel S. (1994) Predicting richness of native, rare, and exotic plants in response to habitat and disturbance variables across a variegated landscape. *Conservation Biology* **8**, 521-31.

McKenzie N., Henderson B. & McDonald W. (2002) Monitoring soil change, principles and practices for Australian conditions. Technical report 18/02. CSIRO Land and Water, Canberra.

McKenzie N., Jacquier D., Isbell R. & Brown K. (2004) *Australian Soils and Landscapes: An Illustrated Compendium*. CSIRO Publishing, Collingwood.

McKenzie N., Ryan P., Fogarty P. & Wood J. (2000) Sampling, measurement and analytical protocols for carbon estimation in soil, litter and coarse woody debris. National Carbon Accounting System Technical Report No. 14. Australian Greenhouse Office, Canberra.

McNeely J. A. & Scherr S. J. (2003) *Ecoagriculture: Strategies to Feed the World and Save Wild Biodiversity*. Island Press, Washington, DC.

McNeill J. R. & Winiwarter V. (2004) Breaking the Sod: Humankind, History and Soil. *Science* **304**, 1627-9.

McRae D., Roth G. & Bange M. (2007) Climate change in cotton catchment communities - A scoping study. Cotton Catchment Communities CRC, Narrabri, NSW.

Mensforth L. J., Thorburn P. J., Tyerman S. D. & Walker G. R. (1994) Sources of water used by riparian *Eucalyptus camaldulensis* overlying highly saline groundwater. *Oecologia* **100**, 21-8.

Metcalfe L., Sivertsen D. P., Tindall D. & Ryan K. M. (2003) Natural vegetation of the New South Wales wheat-belt (Cobar—Nyngan—Gilgandra, Nymagee—Narromine—Dubbo 1:250 000 vegetation sheets). *Cunninghamia* **8**, 253-84.

Milchunas D. G. & Lauenroth W. K. (1993) Quantitative effects of grazing on vegetation and soils over a globle range of environments. *Ecological Monographs* **63**, 327-66.

Milchunas D. G., Lauenroth W. K. & Burke I. C. (1998) Livestock grazing: Animal and plant biodiversity of shortgrass steppe and the relationship to ecosystem function. *OIKOS* **83**, 65-74.

Milchunas D. G., Sala O. E. & Lauenroth W. K. (1988) A generalized model of the effects of grazing by large herbivores on grassland community structure. *The American Naturalist* **132**, 87-106.

Moberg F. & Folke C. (1999) Ecological goods and services of coral reef ecosystems. *Ecological Economics* **29**, 215-33.

Mokany K., Raison R. J. & Prokushkin A. S. (2006) Critical analysis of root:shoot ratios in terrestrial biomes. *Global Change Biology* **12**, 84-96.

Molinar F., Galt D. & Holechek J. (2001) Managing for mulch. *Rangelands* **23**, 3-7.

Mooney H. A. (1972) The carbon balance of plants. *Annual Review of Ecology & Systematics* **3**, 315-46.

Mooney H. A. & Ehrlich P. R. (1997) Ecosystem Services: A Fragmentary History. In: *Natures Services: Societal Dependence on Natural Ecosystems* (ed G. Daily). Island Press, Washington.

Moore R. M. (1975) *Australian Grasslands*. Australian National University Press, Canberra.

Naidoo R., Balmford A., Costanza R., Fischer B., Green R. E., Lehner B., Malcolm T. R. & Ricketts T. H. (2008) Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Science* **105**, 9495-500.

Naidu R., McClure S., Mckenzie N. J. & Fitzpatrick R. W. (1996) Soil solution composition and aggregate stability changes caused by long-term farming at four contrasting sites in South Australia. *Australian Journal of Soil Research* **34**, 511-27.

Naiman R. J., Décamps H. & McClain M. E. (2005) *Riparia: Ecology, Conservation, and Management of Streamside Communities*. Elsevier Inc, San Diego.

Naiman R. J., Décamps H. & Pollock M. (1993) The role of riparian corridors in maintaining regional biodiversity. *Ecological Applications* **3**, 209-12.

Namoi Catchment Management Board (CMB). (2003) Namoi Catchment Blueprint. Department of Land and Water Conservation, Tamworth.

Nelson E., Mendoza G., Regetz J., Polasky S., Tallis H., Cameron D. R., Chan K. M. A., Daily G. C., Goldstein J., Kareiva P. M., Lonsdorf E., Naidoo R., Ricketts T. H. & Shaw M. R. (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment* **7**, 4-11.

Neumann J. (1980) Five letters from and to Hammurapi, King of Babylon (1792-1750 B.C.) on water works and irrigation. *Journal of Hydrology* **47**, 393-7.

New South Wales Scientific Committee. (2004) Coolibah - Black Box Woodland of the northern riverine plains in the Darling Riverine Plains and Brigalow Belt South bioregions - endangered ecological community listing., Sydney.

NLWRA. (2002) Australians and Natural Resource Management 2002. National Land and Water Resources Audit. Land and Water Australia, Canberra.

Norberg J. (1999) Linking nature's services to ecosystems: Some general ecological concepts. *Ecological Economics* **29**, 183-202.

Northcote K. H. (1966) *Atlas of Australian Soils. Explanatory Data for Sheet 3, Sydney-Canberra-Bourke-Armidale Area.* Melbourne University Press, Melbourne.

Northcote K. H. & Skene J. K. M. (1972) *Australian Soils with Saline and Sodic Properties*. CSIRO Publishing.

Noss R. F. & Harris L. D. (1986) Nodes, networks, and MUMs: Preserving diversity at all scales. *Environmental Management* **10**, 299-309.

NSW Scientific Committee. (2005) Myall woodland in the Darling River Plain, Brigalow Belt South, Cobar Peneplain, Murray-Darling Depression, Riverina and South Western Slopes bioregions - endangered ecological community determination - final. (ed DEC NSW), Sydney.

Oades J. M. (1984) Soil organic matter and structural stability: Mechanisms and implications for management. *Plant and Soil* **76**, 319-37.

Oades J. M. (1995) Krasnozems - Organic matter. *Australian Journal of Soil Research* **33**, 43-57.

Odum E. P. (1969) The strategy of ecosystem development. *Science* **164**, 262-70.

Oldeman L. R. (1994) The global extent of soil degradation. In: *Soil Resilience and Sustainable Land Use* (eds D. J. Greenland and I. Szabolcs) pp. 99-118, Budapest.

Olff H. & Ritchie M. E. (1998) Effects of herbivores on grassland plant diversity. *TREE* **13**, 261-5.

Oliver I., Ede A., Hawes W. & Grieve A. (2005) The NSW Environmental Services Scheme: Results for the biodiversity benefits index, lessons learned, and the way forward. *Ecological Management & Restoration* **6**, 197-205.

Oliver I., Jones H. & Schmoldt D. L. (2007) Expert panel assessment of attributes for natural variability benchmarks for biodiversity. *Austral Ecology* **32**, 453-75.

Oliver I., Smith P. L., Lunt I. & Parkes D. (2002) Pre-1750 vegetation, naturalness and vegetation condition: What are the implications for biodiversity conservation. *Ecological Management & Restoration* **3**, 176-8.

Olivier J. G. J., van Aardenne J. A., Dentener F., Ganzeveld L. & Peters J. A. H. W. (2005) Recent trends in global greenhouse gas emissions: Regional trends and spatial distribution of key sources. In: *Proceedings of the Fourth International Symposium of Non-CO_2 Greenhouse Gasses* (ed A. v. Amstel). Millpress, Rotterdam.

Opschoor J. B. (1998) The value of ecosystem services: Whose values? *Ecological Economics* **25**, 41-3.

Orr D. M. (1980a) Effects of sheep grazing *Astrebla* grassland in Central Western Queensland. I Effects of grazing pressure and livestock distribution. *Australian Journal of Agricultural Research* **31**, 797-806.

Orr D. M. (1980b) Effects of sheep grazing *Astrebla* grassland in Central Western Queensland. II Effects of seasonal rainfall. *Australian Journal of Agricultural Research* **31**, 807-20.

Pacala S. W., Hurtt G. C., Baker D., Peylin P., Houghton R. A., Birdsey R. A., Heath L., Sundquist E. T., Stallard R. F., Ciais P., Moorcroft P., Caspersen J. P., Shevliakova E., Moore B., Kohlmaier G., Holland E., Gloor M., Harmon M. E., Fan S. M., Sarmiento J. L., Goodale C. L., Schimel D. & Field C. B. (2001) Consistent land- and atmosphere-based U.S. carbon sink estimates. *Science* **292**, 2316-20.

Palmer G. C. & Bennett A. F. (2006) Riparian zones provide for distinct bird assemblages in forest mosaics of south-east Australia. *Biological Conservation* **130**, 447-57.

Palmer M. A., Ambrose R. F. & Poff N. L. (1997) Ecological theory and community restoration ecology. *Restoration Ecology* **5**, 291-300.

Palmer M. A., Bernhardt E., Chornesky E., Collins S., Dobson A. P., Duke C., Gold B., Jacobson R., Kingsland S., Kranz R., Mappin M., Martinez M. L., Micheli F., Morse J., Pace M., Pascual M., Palumbi S., Reichman O. J., Simons A., Townsend A. & Turner M. (2004) Ecology for a crowded planet. *Science* **304**, 1251-2.

Parkes D. & Lyon P. (2006) Towards a national approach to vegetation condition assessment that meets government investors' needs: A policy perspective. *Ecological Management & Restoration* **7**, S3-S5.

Peasley B. (2000) East Walgett vegetation mapping extant vegetation. NSW Department of Land and Water Conservation, Inverell.

Peel M. C., Finlayson B. L. & McMahon T. A. (2007) Updated world map of the Koppen-Geiger climate classification. *Hydrology and Earth Science Discussions* **4**, 439-73.

Pernes-Debuyser A. & Tessier D. (2004) Soil physical properties affected by long-term fertilization. *European Journal of Soil Science* **55**, 505-12.

Pettit N. E., Froend R. H. & Ladd P. G. (1995) Grazing in remnant woodland vegetation: changes in species composition and life form groups. *Journal of Vegetation Science* **6**, 121-30.

Peverill K. I., Sparrow L. A. & Reuter D. J. (1999) *Soil Analysis, An Interpretation Manual*. CSIRO Publishing, Collingwood.

Pillai-McGarry U. P. P. & Collis-George N. (1990) Laboratory simulation of the surface morphology of self-mulching and non self-mulching vertisols. I. Materials, methods and preliminary results. *Australian Journal of Soil Research* **28**, 129-39.

Pimental D., Stachow U., Takacs D. A., Brubaker H. W., Dumas A. R., Meaney J. J., O'Neil J. A. S., Onsi D. E. & Corzilius D. B. (1992) Conserving biological diversity in agricultural/forestry systems. *BioScience* **42**, 354-62.

Pimentel D., Harvey C., Resosudarmo P., Sinclair K., Kurz D., McNair M., Crist S., Shpritz L., Fitton L., Saffouri R. & Blair R. (1995) Environmental and economic costs of soil erosion and conservation benefits. *Science* **267**, 1117-23.

Pimentel D., McNair S., Janecka J., Wightman J., Simmonds C., O'Connell C., Wong E., Russel L., Zern J., Aquino T. & Tsomondo T. (2001) Economic and environmental threats of alien plant, animal, and microbe invasions. *Agriculture, Ecosystems & Environment* **84**, 1-20.

Piper S. D. & Catterall C. P. (2003) A particular case and a general pattern: Hyperaggressive behaviour by one species may mediate avifaunal decreases in fragmented Australian forests. *OIKOS* **101**, 602-14.

Possingham H. P. & Wilson K. A. (2005) Turning up the heat on hotspots. *Nature* **436**, 919.

Post W. M. & Kwon K. C. (2000) Soil carbon sequestration and land-use change: Processes and potential. *Global Change Biology* **6**, 317-27.

Pratley J. (2003) *Principles of Field Crop Production*. Oxford University Press, New York.

Prebble R. E. (1987) Effect of cultivation on aggregate stability, micro-aggregation and organic carbon in vertisols. CSIRO Division of Soils Report No. 91. CSIRO, Canberra.

Prescott J. J. (1944) A Soil Map of Australia. Bulletin No. 177. CSIRO Publishing.

Pressey R. L., Humphries C. J., Margules C. R., Vane-Wright R. I. & Williams P. H. (1993) Beyond opportunism: Key principles for systematic reserve selection. *TRENDS in Ecology and Evolution* **8**, 124-8.

Pressey R. L. & Nicholls A. O. (1989) Efficiency in conservation evaluation: Scoring versus iterative approaches. *Biological Conservation* **50**, 199-218.

Pressland A. J. (1984) Productivity and management of western Queensland's rangeland. *Australian Rangeland Journal* **6**.

Prober S. M. & Thiele K. R. (1995) Conservation of the grassy white box woodlands: Relative contributions of size and disturbance to floristic composition and diversity of remnants. *Australian Journal of Botany* **43**, 349-66.

Prober S. M. & Thiele K. R. (2004) Floristic patterns along an east-west gradient in grassy box woodlands of central New South Wales. *Cunninghamia* **8**, 306-25.

Prober S. M., Thiele K. R. & Lunt I. D. (2002) Identifying ecological barriers to restoration in temperate grassy woodlands: Soil changes associated with different degradation states. *Australian Journal of Botany* **50**, 699-712.

Prober S. M., Thiele K. R., Lunt I. D. & Koen T. B. (2005) Restoring ecological function in temperate grassy woodlands: Manipulating soil nutrients, exotic annuals and native perennial grasses through carbon supplements and spring burns. *Journal of Applied Ecology* **42**, 1073-85.

Proctor W., Cork S., Langridge J., Langston A., Abel N., Howden M., Anderies M., Parry R. & Shelton D. (2002) Assessing ecosystems services in Australia. In: *7th Biennial Conference of the International Society of Ecological Economics,* Sousse, Tunisia.

Prosser I. P., Rutherfurd I. D., Olley J. M., Young W. J., Wallbrink P. J. & Moran C. J. (2001) Large-scale patterns of erosion and sediment transport in river networks, with examples from Australia. *Marine and Freshwater Research* **52**, 81-99.

R Development Core Team. (2009) A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Redman R. E. (1992) Primary productivity. In: *Ecosystems of the World: Introduction and Western Hemisphere. Volume 8A* (ed R. T. Coupland) pp. 75-94. Elsevier, Amsterdam.

Rees R. M., Bingham I. J., Baddeley J. A. & Watson C. A. (2005) The role of plants and land management in sequestering soil carbon in temperate arable and grassland ecosystems. *Geoderma* **128**, 130-54.

Reid J. R. W. (1999) Threatened and declining birds in the New South Wales sheep-wheat belt: 1. Diagnosis, characteristics and management. Consultancy report to NSW National Parks and Wildlife Service, CSIRO Wildlife and Ecology, Canberra.

Reid N., Karanja F. & Thompson D. (2006) Ecosystem services and biodiversity indicators. In: *13th Australian Cotton Conference*, Gold Coast.

Reid N. & Landsberg J. (2000) Tree decline in agricultural landscapes: what we stand to lose. In: *Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Managmement and Restoration* (eds R. J. Hobbs and C. J. Yates) pp. 127-66. Surrey Beatty & Sons, Chipping Norton.

Reid N., Nadolny C., Banks V., O'Shae G. & Jenkins B. (2007) Causes of eucalypt tree decline in the Namoi Valley, NSW. Final Report to Land and Water Australia on Project UNE 42. Ecosystem Management, University of New England, Armidale.

Reid N., O'Shea G. & Silberbauer L. (2003) A review of biodiversity research in the Australian cotton industry: Final report to the Cotton Research and Development Corporation. University of New England, Armidale.

Rengasamy P. (2002) Transient salinity and subsoil constraints to dryland farming in Australian sodic soils: An overview. *Australian Journal of Experimental Agriculture* **42**, 351-61.

Rengasamy P. & Churchman G. J. (1999) Cation exchange capacity, exchangeable cations and sodicity. In: *Soil Analysis: An Interpretation Manual* (eds K. I.

Peverill, L. A. Sparrow and D. J. Reuter) pp. 147-57. CSIRO Publishing, Collingwood.

Rengasamy P., Greene R. S. B., Ford G. W. & Mehanni A. H. (1984) Identification of dispersive behaviour and the management of red-brown earths. *Australian Journal of Soil Research* **22**, 413-31.

Rengasamy P. & Olsson K. A. (1991) Sodicity and soil structure. *Australian Journal of Soil Research* **29**, 935-52.

Ribaudo M. O., Hoag D. L., Smith M. E. & Heimlich R. (2001) Environmental indices and the politics of the conservation reserve program. *Ecological Indicators* **1**, 11-20.

Riley S. J. & Taylor G. (1978) The geomorphology of the upper Darling River system with special reference to the present fluvial system. *Royal Society of Victoria* **90**, 89-102.

Ringrose S., Matheson W., Matlala C. J. S. & O'Neil T. (1994) Vegetation spectral reflectance along a north-south vegetation gradient in northern Australia. *Journal of Biogeography* **21**, 33–47.

Rodhe H. (1990) A comparison of the contribution of various gases to the greenhouse effect. *Science* **248**, 1217-9.

Rodriguez J. P., Beard J., T.D., Bennett E. M., Cumming G. S., Cork S. J., Agard J., Dobson A. P. & Peterson G. D. (2006) Trade-offs across space, time, and ecosystem services. *Ecology and Society* **11**, 28 [online].

Rosewell C. J. & Loch R. J. (2002) Estimation of the RUSLE soil erodibility factor. In: *Soil Physical Measurement and Interpretation for Land Evaluation* (eds N. McKenzie, K. Coughlan and H. Cresswell). CSIRO Publishing, Collingwood.

Rounsevell M. D. A., Dawson T. P. & Harrison P. A. (2010) A conceptual framework to assess the effects of environmental change on ecosystem services. *Biodiversity and Conservation* **online early**.

Ruiz-Jaen M. C. & Aide T. M. (2005) Restoration success: How is it being measured? *Restoration Ecology* **13**, 569-77.

Running S. W., Loveland T. R. & Pierce L. (1994) A vegetation classification logic based on remote sensing for use in global biogeochemical models. *Ambio* **23**, 77-81.

Salt D., Lindenmayer D. & Hobbs R. (2004) Trees and biodiversity: A guide for Australian farm forestry. Rural Industries Research and Developement Corporation, Canberra.

Sampson A. W. (1917) Succession as a factor in range management. *Journal of Range Management* **15**, 593-6.

Satellite Imaging Corporation. (2003) SPOT 5 Satellite Imagery and Sensor Characteristics. http://www.satimagingcorp.com/satellite-sensors/spot-5.html.

Saunders D. A., Hobbs R. J. & Margules C. R. (1991) Biological consequences of ecosystem fragmentation: A review. *Conservation Biology* **5**, 18-32.

Scanlan J. C. & Burrows W. H. (1990) Woody overstorey impact on herbaceous understorey in *Eucalyptus spp.* communities in central Queensland. *Austral Ecology* **15**, 191-7.

Schimel D. S., House J. I., Hibbard K. A., Bousquet P., Ciais P., Peylin P., Braswell B. H., Apps M. J., Baker D., Bondeau A., Canadell J., Churkina G., Cramer W., Denning A. S., Field C. B., Friedlingstein P., Goodale C., Heimann M., Houghton R. A., Melillo J. M., Moore B., Murdiyarso D., Noble I., Pacala S. W., Prentice I. C., Raupach M. R., Rayner P. J., Scholes R. J., Steffen W. L. & Wirth C. (2001) Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. *Nature* **414**, 169-72.

Schlich W. (1896) *A manual of forestry*. Bradbury, Agnew, London.

Scholes R. J. (2003) Convex relationships in ecosystems containing mixtures of trees and grass. *Environmental and Resource Economics* **26**, 559-74.

Scholes R. J. & Archer S. R. (1997) Tree-grass interations in savannas. *Annual Review of Ecology and Systematics* **28**, 517-44.

Scholes R. J. & Hall D. O. (1996) The carbon budget of tropical savannas, woodlands and grasslands. In: *Global change: Effects on Coniferous Forests and Grasslands* (eds A. I. Breymeyer, D. O. Hall, J. M. Melillo and G. I. Ågren) pp. 69-100. John Wiley & Sons, Chichester.

Schulze E. D., Wirth C. & Heimann M. (2000) Climate change: Managing forests after Kyoto. *Science* **289**, 2058-9.

Schwartz G. (1978) Estimating the dimension of a model. *Annals of Statistics* **6**, 461-4.

Schwartz M. W., Brigham C. A., Hoeksema J. D., Lyons K. G., Mills M. H. & van Mantgen P. J. (2000) Linking biodiversity to ecosystem function: implications for conservation ecology. *Oecologia* **122**, 297-305.

Selby M. J. (1993) Mass wasting of soils. In: *Hillslope Materials and Processes. Second Edition* (ed M. J. Selby) pp. 249-98. Oxford University Press, Oxford.

Sharma M. L. & Tongway D. J. (1973) Plant induced soil salinity patterns in two saltbush (*Atriplex* spp.) communities. *Journal of Range Management* **26**, 121-5.

Silburn D. M., Waters D. K., Connolly R. D., Simpson B. W. & Kennedy I. R. (1999) Techniques for stabilising soil erosion on cotton farms. In: *Minimising the impact of pesticides in the riverine environment: key findings from research with the cotton industry: LWRRDC Occasional Paper 23/98* (eds N. J. Schofield and V. E. Edge). Land and Water Resources Reseach and Development Corporation, Canberra.

Sitch S., Smith B., Prentice I. C., Arneth A., Bondeau A., Cramer W., Kaplan J. O., Levis S., Lucht W., Sykes M. T., Thonicke K. & Venevsky S. (2003) Evaluation of ecosystem dynamics, plant geography and terrestrial carbon cycling in the LPJ dynamic global vegetation model. *Global Change Biology* **9**, 161-85.

Sivertsen D. (1994) The native vegetation crisis in the wheatbelt of NSW. *Search* **25**, 5-8.

Sivertsen D. & Metcalfe L. (2001) Northern wheatbelt vegetation mapping. NSW National Parks and Wildlife Service, Hurstville.

Smith G. D. (1984) Soil constituent and prehistory effects on aggregate porosity in cracking clays. In: *The Properties and Utilization of Cracking Clay Soils* (eds J. W. McGarity, E. H. Hoult and H. B. So). University of New England, Armidale.

Snowdon P., Eamus D., Gibbons P., Khanna P., Keith H., Raison J. & Kirschbaum M. (2000) Synthesis of allometrics, review of root biomass and design of future woody biomass sampling strategies. Technical report no. 17. Australian Greenhouse Office, Canberra.

Snowdon P., Ryan P. & Raison J. (2005) Review of C:N ratios in vegetation, litter and soil under Australian native forests and plantations. Technical Report No. 24. Australian Greenhouse Office, Canberra.

Spain A. V. & Hodgen M. J. (1994) Changes in the composition of sugarcane harvest residues during decomposition as a surface mulch. *Biology and Fertility of Soils* **3**, 225-31.

Spain A. V., Isbell R. & Probert M. E. (1983) Soil organic matter. In: *Soils: An Australian viewpoint* (ed CSIRO Division of Soils) pp. 551-63. Academic Press, London.

SPCC. (1980) Namoi Environment Study. State Pollution Control Commission, New South Wales.

Specht R. L. (1970) Vegetation. In: *The Australian Environment* (ed G. W. Leeper) pp. 44-67. CSIRO and Melbourne University Press, Melbourne.

Stace H. C. T., Hubble G. D., Brewer R., Northcote K. H., Sleeman J. R., Mulcahy M. J. & Hallsworth E. G. (1968) *A Handbook of Australian Soils*. Rellim Technical Publications for CSIRO and International Society of Soil Science, Glenside.

Stannard M. E. & Kelly I. D. (1977) The Irrigation Potential of the Lower Namoi Valley. Water Resources Commission, Sydney.

Steenekamp S. J. & Bosch O. J. H. (1995) Construction and evaluation of condition assessment and grazing capacity models for use in grazing management in the Eastern Mixed Bushveld of southern Africa. *Journal of Arid Environments* **30**, 351-60.

Stephens C. G. (1961) The Soil Landscapes of Australia. Publication No. 18. CSIRO Soils Division, Canberra.

Stokes C. & Howden M. (2008) An overview of climate change adaption in Australian primary industries - impacts, options and priorities. Report prepared for the National Climate Change Research Strategy for Primary Industries. CSIRO.

Strand H., Höft R., Strittholt J., Miles L., Horning N., Fosnight E. & Turner W. (2007) Sourcebook on Remote Sensing and Biodiversity Indicators Coverage of Protected Areas. CBD Technical Series No. 32. Secretariat of the Convention on Biological Diversity, Montréal.

Sumner M. E. (1993) Sodic soils: New perspectives. *Australian Journal of Soil Research* **31**, 683-750.

Swift M. J., Heal O. W. & Anderson J. M. (1979) *Decomposition in Terrestrial Ecosystems*. Blackwell Scientific Publications, Oxford.

Swift M. J., Izac A.-M. N. & van Noordwijk M. (2004) Biodiversity and ecosystem services in agricultural landscapes—are we asking the right questions? *Agriculture, Ecosystems and Environment* **104**, 113-34.

Swinton S. M., Lupi F., Robertson G. P. & Hamilton S. K. (2007) Ecosystem services and agriculture: Cultivatin agricultural ecosystems for diverse benefits. *Ecological Economics* **64**, 245-52.

Symstad A. J., Chapin F. S., Wall D. H., Gross K. L., Huenneke L. F., Mittelbach G. G., Peters D. P. C. & Tilman D. (2003) Long-term and large-scale perspectives on the relationship between biodiversity and ecosystem functioning. *BioScience* **53**, 89-98.

Taddese G., Saleem M. A. M. & Ayalneh W. (2002) Effect of livestock grazing on physical properties of a cracking and self-mulching vertisol. *Australian Journal of Experimental Agriculture* **42**, 129-33.

Taylor B. & Ganf G. G. (2005) Comparative ecology of two co-occurring floodplain plants: The native *Sporobolus mitchellii* and the exotic *Phyla canescens*. *Marine and Freshwater Research* **56**, 431-40.

Thackway R. & Cresswell I. D. (1995) An Interim Biogeographic Regionalisation for Australia. Version 6.1. Department of Environment and Heritage, Canberra.

Thackway R. & Lesslie R. (2006) Repoting vegetation condition using the Vegetation Assets, States and Transitions (VAST) framework. *Ecological Management and Restoration* **7**, S53-S62.

Thoms M., Norris R., Harris J., Williams D. & Cottingham P. (1999) Environmental scan of the Namoi River Valley. Prepared for the Department of Land and Water Conservation and the Namoi River Management Committee.

Thorburn P. J., Hatton T. J. & Walker G. R. (1993) Combining measurements of transpiration and stable isotopes of water to determine groundwater discharge forests. *Journal of Hydrology* **150**, 563-87.

Thorburn P. J., Mensforth L. J. & Walker G. R. (1994) Reliance of creek-side river red gums on creek water. *Australian Journal of Marine & Freshwater Research* **45**, 1439-43.

Thorburn P. J. & Walker G. R. (1994) Variations in stream water uptake by *Eucalyptus camaldulensis* with differing access to stream water. *Oecologia* **100**, 293-301.

Tilman D. (1999) Global environmental impacts of agricultural expansion: The need for sustainable and efficient practices. *Proceedings of the National Academy of Sciences of the United States of America* **96**, 5995-6000.

Tilman D., Wedin D. & Knops J. (1996) Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* **379**, 718-20.

Tisdall J. M. & Oades J. M. (1982) Organic matter and water stable aggregates in soil. *Journal of Soil Science* **33**, 141-63.

Toman M. (1998) Why not to calculate the value of the world's ecosystem services and natural capital. *Ecological Economics* **25**, 57-60.

Tongway D. & Hindley N. L. (2004) Landscape Function Analysis: Procedures for Monitoring and Assessing Landscapes. CSIRO Sustainable Ecosystems, Canberra.

Tongway D. & Ludwig J. (1997) The nature of landscape dysfunction in rangelands. In: *Landscape Ecology Function and Management* (eds J. Ludwig, D. Tongway, D. Freudenberger, J. Noble and K. Hodgkinson). CSIRO Publishing, Canberra.

Tongway D. J. & Ludwig J. A. (1994) Small-scale resource heterogeneity in semi-arid landscapes. *Pacific Conservation Biology* **1**, 201-8.

Tongway D. J. & Whitford W. (2002) *Desertification and soil processes in rangelands*. CABI Publishing, Wallingford.

Tothill J. C., Hargreaves J. N. G. & Jones R. M. (1978a) BOTANAL - A comprehensive sampling and computing procedure for estimating yield and composition. 1. Field sampling. CSIRO Brisbane.

Tothill J. C., Hargreaves J. N. G. & Jones R. M. (1978b) A comprehensive sampling and computing procedure for estimating pasture yield and composition 1. Field Sampling Tropical Agronomy Technical Memorandum Number 8. CSIRO Division of Tropical Crops and Pastures, Brisbane.

Towler R. J. (1984) Broad-based banks and their place in agriculture. In: *The Properties and Utilisation of Cracking Clay Soils* (eds J. W. McGarity, E. H. Hoult and H. B. So). University of New England, Armidale.

Trémont R. M. & McIntyre S. (1994) Natural grassy vegetation and native forbs in temperate Australia: Structure, dynamics and life-histories. *Australian Journal of Botany* **42**, 641-58.

Tscharntke T., Klein A. M., Kruess A., Steffan-Dewenter I. & Thies C. (2005) Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecology Letters* **8**, 857-74.

Tucker B. M. (1954) The determination of exchangeable calcium and magnesium in carbonate soils. *Australian Journal of Soil Research* **5**, 706-15.

Turner R. K., Adger W. N. & Brouwer R. (1998) Ecosystem services value, research needs, and policy relevance: A commentary. *Ecological Economics* **25**, 61-5.

Turner R. K., Paavola J., Cooper P., Farber S., Jessamy V. & Georgiou S. (2003) Valuing nature: Lessons learned and future research directions. *Ecological Economics* **46**, 493-510.

Turner W. R., Brandon K., Brooks T. M., Costanza R., da Fonseca G. A. B. & Portela R. (2007) Global conservation of biodiversity and ecosystem services. *BioScience* **57**, 868-73.

United Nations. (1998) Kyoto Protocol to the United Nations Framework Convention on Climate Change. http://unfccc.int/resource/docs/convkp/kpeng.pdf.

Usher M. B. (1986) *Wildlife Conservation Evaluation*. Chapman and Hall, London.

Vogelmann J. E., Sohl T. & Howard S. M. (1998) Regional characterization of land cover using multiple sources of data. *Photogrammetric Engineering and Remote Sensing* **64**, 45–57.

Walker B., Kinzig A. & Langridge J. (1999) Plant attribute diversity, resilience, and ecosystem function: the nature and significance of dominant and minor species. *Ecosystems* **2**, 95-113.

Walker B. H. (1992) Biodiversity and Ecological Redundancy. *Conservation Biology* **6**, 18-23.

Walker J. & Reuter D. J. (1996) *Indicators of Catchment Health: A Technical Perspective*. CSIRO, Collingwood.

Wallace K. J. (2007) Classification of ecosystem services: Problems and solutions. *Biological Conservation* **139**, 235-46.

Wallington T., Hobbs R. J. & Moore S., A. (2005) Implications of current ecological thinking for biodiversity conservation: A review of the salient issues. *Ecology and Society* **10**, 15-31.

Wang G. (1998) Is height of dominant trees at a reference diameter an adequate measure of site quality? *Forest Ecology and Management* **112**, 49-54.

Ward W. T. (1999) Soils and landscapes near Narrabri and Edgeroi, NSW, with data anallysis using fuzzy k-means. CSIRO Land and Water Technical Report. CSIRO, Canberra.

Water Resources Commission. (1980) Floods in the Namoi Valley, with particular reference to 1955, 1962, 1964, 1971, 1974, 1976. Flood Information Reports. New South Wales Water Resources Commission, Sydney.

Watson R. T., Noble I. R., Bolin B., Ravindranath N. H., Verardo D. J. & Dokken D. J. (2000) Land use, land-use change, and forestry. Special report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge.

Weltzin J. F. & Coughenour M. B. (1990) Savanna tree influence on understorey vegetation and soil nutrients in northwest Kenya. *Journal of Vegetation Science* **1**, 325-34.

Westoby M., Walker B. & Noy-Meir I. (1989) Opportunistic management for rangelands not at equilibrium. *Journal of Range Management* **42**, 266-74.

Wijesuriya S. & Hocking C. (1999) Why do weeds grow when you dig up native grasslands? The effects of physical disturbance on available nutrients, mineralisation, and weed invasion in grassland soil. In: *Down to grass roots:*

Proceedings of a conference on the management of grassy ecosystems (ed V. C. C. Hocking) pp. 31-7. Department of Conservation and Natural Resources and Parks Victoria, Victoria University St. Albans, 9 and 10 July 1998.

Williams A. (2005) Managing the environmental impacts of cotton-growing - an Australian perspective. In: *64th ICAC plenary meeting*, Liverpool, UK.

Williams J., Saunders D. & Colloff M. (2001) Using biodiversity standards and targets to assist with the mitigation of land and water degradation. In: *WWF Australia Murray-Darling Basin 2051: Setting the Vision, Biodiveristy Workshop*, Canberra.

Wilson A. D. (1990) The effect of grazing on Australian ecosystems. In: *Proceedings of the Ecological Society of Australia. Volume 16* pp. 235-44, Symposium held in Geraldton, Western Australia.

Wilson B. R., Growns I. & Lemon J. (2008) Land-use effects on soil carbon and other soil properties on the NW slopes of NSW: Implications for soil condition assessment. *Australian Journal of Soil Research* **45**, 199-205.

Wilson G. W. T., Rice C. W., Rillig M. C., Springer A. & Hartnett D. C. (2009) Soil aggregation and carbon sequestration are tightly correlated with the abundance of arbuscular micorrhizal fungi: Results from long-term field experiments. *Ecology Letters* **12**.

Wilson K., Pressey R. L., Newton A., Burgan M., Possingham H. & Weston C. (2005) Measuring and incorporating vulnerability into conservation planning. *Environmental Management* **35**, 527-43.

Wilson K. A., Underwood E. C., Morrison S. A., Klausmeyer K. R., Murdoch W. W., Reyers B., Wardell-Johnson G., Marquet P. A., Rundel P. W., McBride M. F., Pressey R. L., Bode M., Hoekstra J. M., Andelman S., Looker M., Rondinini C., Kareiva P., Shaw M. R. & Possingham H. P. (2007) Conserving biodiversity efficiently: What to do, where, and when. *PLoS Biol* **5**.

Windle J., Rolf J., McCosker J. & Lingard A. (2009) A conservation auction for landscape linkage in the southern Desert Uplands, Queensland *Rangeland Journal* **31**, 127-35.

Woinarski J. C. Z., Brock C., Armstrong B. M., Hempel C., Cheal D. & Brennan K. (2000) Bird distribution in riparian vegetation in the extensive natural landscape of Australia's tropical savanna: A broad-scale survey and analysis of a distributional data base. *Journal of Biogeography* **27**, 843-68.

Woodgate P. & Black P. (1988) Forest cover changes in Victoria 1869-1987. Remote Sensing Group, Lands and Forests Division. Department of Conservation, Forests and Lands, Melbourne.

Xiong S. & Nilsson C. (1997) Dynamics of leaf litter accumulation and its effects on riparian vegetation: A review. *The Botanical Review* **63**, 240-64.

Yates C. J., Norton D. A. & Hobbs R. J. (2000) Grazing effects on plant cover, soil and microclimate in fragmented woodlands in south-western Australia: Implications for restoration. *Austral Ecology* **25**, 36-47.

Yates W. J. (1972) Factors Affecting the Surface Structural Stability of Clay Soils in the Moree Area, NSW. University of New England, Armidale.

Yates W. J. & McGarity J. W. (1984) Structural instability in grey clay surface soils. In: *The Properties and Utilisation of Cracking Clay Soils* (eds J. W. McGarity, E. H. Hoult and H. B. So). University of New England, Armidale.

Yatskov M., Harmon M. E. & Krankina O. N. (2003) A chronosequence of wood decomposition in the boreal forests of Russia. *Canadian Journal of Forest Research* **33**, 1211-26.

Young R., Wilson B. R., McLeod M. & Alston C. (2005) Carbon storage in the soils and vegetation of contrasting land uses in northern New South Wales, Australia. *Australian Journal of Soil Research* **43**, 21-31.

Young R. R., Wilson B., Harden S. & Bernardi A. (2009) Accumulation of soil carbon under zero tillage cropping and perennial vegetation on the Liverpool Plains, eastern Australia. *Australian Journal of Soil Research* **47**, 273-85.

Zhang W., Ricketts T. H., Kremen C., Carney K. & Swinton S. M. (2007) Ecosystem services and dis-services to agriculture. *Ecological Economics* **64**, 253-60.

Zhou G., Liu S., Li Z., Zhang D., Tang X., Zhou C., Yan J. & Mo J. (2006) Old-growth forests can accumulate carbon in soils. *Science* **314**, 1417-8.

Zonneveld I. S. (1988) Survey approaches. In: *Vegetation Mapping* (eds A. W. Küchler and I. S. Zonneveld) pp. 203–8. Kluwer Academic Publishers, Dordrecht.

Appendix 1: Native, long-lived perennial plant species recorded during surveys of the lower Namoi floodplain.

				Average projected foliage cover (%) in each vegetation type				
Species	**Family**	**Life-form**	**Number of occurrences**	**River Red Gum**	**Coolibah**	**Myall**	**Black Box**	**Grassland**
Abutilon oxycarpum	Malvaceae	forb	1	0.000	0.001	0.000	0.000	0.000
Acacia pendula	Fabaceae - Mimosoideae	shrub	8	0.000	0.294	17.167	0.000	0.001
Acacia salicina	Fabaceae - Mimosoideae	shrub	5	3.078	0.118	0.000	0.000	0.000
Acacia stenophylla	Fabaceae - Mimosoideae	shrub	15	3.847	1.943	0.000	0.000	0.029
Alectryon oleofolius	Sapindaceae	shrub	18	0.002	0.531	0.003	0.005	0.009
Apophyllum anomalum	Capparidaceae	shrub	5	0.000	0.000	0.168	0.003	0.002
Aristida calycina	Poaceae	grass	1	0.000	0.000	0.002	0.000	0.000
Aristida leptopoda	Poaceae	grass	2	0.000	0.000	0.000	0.000	0.001
Aristida ramosa	Poaceae	grass	1	0.000	0.006	0.000	0.000	0.000
Asperula conferta	Rubiaceae	forb	15	1.016	0.418	0.000	0.000	0.179
Astrebla lappacea	Poaceae	grass	10	0.000	0.029	0.667	0.500	6.883
Atalaya hemiglauca	Sapindaceae	shrub	3	0.000	0.001	0.000	0.003	0.001
Atriplex semibaccata	Chenopodiaceae	chenopod	19	0.000	0.177	0.750	1.500	0.242
Austrodanthonia bipartita	Poaceae	grass	6	0.000	0.178	0.002	0.000	0.235
Austrostipa aristiglumis	Poaceae	grass	7	0.385	0.059	0.002	0.000	2.413
Austrostipa scabra subsp. *scabra*	Poaceae	grass	3	0.000	0.000	0.133	0.000	0.001
Austrostipa verticillata	Poaceae	grass	2	0.002	0.000	0.000	0.000	0.000
Boerhavia dominii	Nyctaginaceae	forb	48	0.265	0.323	0.502	0.450	0.296
Bothriochloa bladhii	Poaceae	grass	1	0.004	0.000	0.000	0.000	0.000
Bothriochloa macra	Poaceae	grass	2	0.008	0.000	0.000	0.000	0.000
Brachyscome dentata	Asteraceae	forb	2	0.000	0.001	0.000	0.000	0.000
Calostemma purpureum	Asteraceae	forb	3	0.155	0.000	0.000	0.000	0.000
Calotis erinacea	Asteraceae	forb	1	0.000	0.001	0.000	0.000	0.000
Calotis scabiosifolia var. *scabiosifolia*	Asteraceae	forb	9	0.000	0.001	0.003	0.008	0.007
Calotis scapigera	Asteraceae	forb	5	0.000	0.092	0.000	0.000	0.000
Capparis lasiantha	Capparidaceae	shrub	5	0.000	0.061	0.000	0.000	0.001
Capparis mitchellii	Capparidaceae	shrub	1	0.000	0.000	0.000	0.000	0.001
Carex inversa	Cyperaceae	graminoid	18	1.435	0.148	0.000	0.000	0.089
Casuarina cristata	Casurinaceae	tree	6	0.000	0.648	0.002	0.000	0.000

Chamaesyce drummondii	Euphorbiaceae	forb	19	0.155	0.004	0.007	0.010	0.002
Chloris truncata	Poaceae	grass	8	0.000	0.071	0.000	0.125	0.072
Chloris ventricosa	Poaceae	grass	2	0.077	0.118	0.000	0.000	0.000
Commelina cyanea	Commelinaceae	forb	5	0.233	0.000	0.000	0.000	0.000
Convolvulus erubescens	Convolvulaceae	forb	2	0.000	0.000	0.000	0.000	0.030
Cullen tenax	Fabaceae - Faboideae	forb	3	0.001	0.000	0.000	0.000	0.001
Cynodon dactylon	Poaceae	grass	10	0.214	0.006	0.000	0.000	0.001
Cyperus bifax	Cyperaceae	graminoid	2	0.085	0.000	0.000	0.000	0.000
Cyperus gracilis	Cyperaceae	graminoid	2	0.002	0.000	0.000	0.000	0.000
Cyperus lucidus	Cyperaceae	graminoid	21	0.778	0.238	0.000	0.003	0.267
Dichanthium sericeum subsp. *sericeum*	Poaceae	grass	9	0.769	0.059	0.000	0.000	0.971
Dichondra sp. A	Convolvulaceae	forb	2	0.078	0.000	0.000	0.000	0.000
Digitaria coenicola	Poaceae	grass	1	0.000	0.000	0.000	0.000	0.001
Digitaria divaricatissima	Poaceae	grass	4	0.000	0.000	0.002	0.125	0.118
Einadia hastata	Chenopodiaceae	chenopod	4	0.038	0.294	0.833	0.000	0.006
Eleocharis plana	Cyperaceae	graminoid	6	1.162	0.059	0.000	0.000	0.001
Elymus plurinervis	Poaceae	grass	3	0.001	0.000	0.000	0.000	0.119
Enchylaena tomentosa	Chenopodiaceae	chenopod	5	0.000	0.002	0.183	0.003	0.000
Enteropogon acicularis	Poaceae	grass	34	0.463	2.765	2.833	0.253	6.561
Eragrostis setifolia	Poaceae	grass	6	0.000	0.235	0.018	0.000	0.001
Eremophila bignoniiflora	Myoporaceae	shrub	9	0.154	0.413	0.008	3.000	0.000
Eremophila debilis	Myoporaceae	forb	10	0.469	0.184	0.002	0.003	0.001
Eremophila maculata	Myoporaceae	shrub	7	0.000	0.001	0.335	0.175	0.001
Eremophila mitchellii	Myoporaceae	shrub	2	0.000	0.000	0.167	0.250	0.000
Eriochloa crebra	Poaceae	grass	1	0.004	0.000	0.000	0.000	0.000
Eriochloa procera	Poaceae	grass	5	0.000	0.595	0.167	0.000	0.006
Eucalyptus camaldulensis	Myrtaceae	tree	16	14.000	0.882	0.000	0.000	0.001
Eucalyptus coolabah	Myrtaceae	tree	32	2.923	14.294	0.002	1.500	0.532
Eucalyptus largiflorens	Myrtaceae	tree	8	0.000	0.000	2.000	10.500	0.035
Eucalyptus populnea	Myrtaceae	tree	2	0.001	0.000	0.000	0.000	0.001
Euchiton sphaericus	Asteraceae	forb	15	0.039	0.002	0.083	0.005	0.033
Geranium solanderi var. *solanderi*	Geraniaceae	forb	1	0.000	0.000	0.000	0.000	0.001
Glycine tabacina	Fabaceae - Faboideae	forb	4	0.017	0.059	0.000	0.000	0.000

Goodenia fascicularis	Goodeniaceae	forb	27	0.002	0.461	0.008	0.255	0.003
Hibiscus brachysiphonius	Malvaceae	forb	5	0.000	0.001	0.002	0.053	0.001
Jasminum suavissimum	Oleaceae	forb	3	0.027	0.000	0.000	0.000	0.000
Juncus aridicola	Juncaceae	graminoid	3	0.002	0.000	0.000	0.000	0.001
Leiocarpa leptolepis	Asteraceae	forb	2	0.000	0.000	0.000	0.000	0.030
Leiocarpa tomentosa	Asteraceae	forb	4	0.000	0.030	0.083	0.000	0.001
Leptochloa ciliolata	Poaceae	grass	3	0.392	0.000	0.000	0.000	0.001
Leptochloa digitata	Poaceae	grass	3	0.002	0.059	0.000	0.000	0.000
Leptochloa divaricatissima	Poaceae	grass	15	0.008	1.130	0.002	0.000	1.182
Lomandra longifolia	Lomandraceae	graminoid	2	0.808	0.000	0.000	0.000	0.000
Maireana aphylla	Chenopodiaceae	chenopod	2	0.000	0.118	0.033	0.000	0.000
Maireana coronata	Chenopodiaceae	chenopod	3	0.000	0.000	0.017	0.003	0.118
Maireana decalvans	Chenopodiaceae	chenopod	8	0.015	0.062	0.350	0.003	0.001
Marsilea drummondii	Marsileaceae	forb	23	0.241	0.151	0.003	0.013	0.061
Melaleuca bracteata	Myrtaceae	shrub	1	0.001	0.000	0.000	0.000	0.000
Melia azedarach	Meliaceae	tree	1	0.001	0.000	0.000	0.000	0.000
Muehlenbeckia florulenta	Polygonaceae	shrub	18	1.900	1.648	0.085	1.000	0.059
Myoporum montanum	Myoporaceae	shrub	4	0.002	0.000	0.000	0.000	0.001
Neptunia gracilis	Fabaceae - Mimosoideae	forb	9	0.000	0.036	0.000	0.000	0.026
Panicum buncei	Poaceae	grass	2	0.000	0.001	0.000	0.000	0.001
Panicum decompositum	Poaceae	grass	21	0.077	0.089	0.502	0.250	2.848
Panicum queenslandicum var. *queenslandicum*	Poaceae	grass	4	0.000	0.059	0.000	0.000	3.559
Paspalidium globoideum	Poaceae	grass	4	0.002	0.000	0.000	0.000	0.059
Paspalidium gracile	Poaceae	grass	26	3.502	0.872	2.002	3.005	0.500
Paspalidium jubiflorum	Poaceae	grass	32	5.965	4.718	0.333	0.000	0.735
Paspalum distichum	Poaceae	grass	1	0.154	0.000	0.000	0.000	0.000
Persicaria prostrata	Polygonaceae	forb	1	0.000	0.000	0.000	0.000	0.001
Phyllanthus virgatus	Phyllanthaceae	forb	5	0.001	0.008	0.000	0.000	0.000
Poa labillardierei var. *labillardierei*	Poaceae	grass	3	0.039	0.000	0.000	0.000	0.001
Polymeria longifolia	Convolvulaceae	forb	13	0.002	0.000	0.250	0.425	0.366
Ptilotus semilanatus	Amaranthaceae	forb	6	0.000	0.029	0.250	0.125	0.059
Ranunculus lappaceus	Ranunculaceae	forb	1	0.001	0.000	0.000	0.000	0.000
Rhagodia spinescens	Chenopodiaceae	chenopod	3	0.001	0.059	0.167	0.000	0.000

Rostellularia adscendens var. *adscendens*	Acanthaceae	forb	6	0.000	0.001	0.033	0.503	0.029
Rumex brownii	Polygonaceae	forb	18	1.207	0.002	0.000	0.000	0.089
Sclerolaena bicornis var. *horrida*	Chenopodiaceae	chenopod	9	0.000	0.588	3.333	2.628	1.176
Sclerolaena birchii	Chenopodiaceae	chenopod	10	0.539	0.118	0.002	0.000	0.118
Sclerolaena calcarata	Chenopodiaceae	chenopod	13	0.000	0.121	1.333	1.875	0.088
Senecio quadridentatus	Asteraceae	forb	1	0.000	0.001	0.000	0.000	0.000
Sida ammophila	Malvaceae	forb	4	0.002	0.000	0.000	0.005	0.001
Sida corrugata	Malvaceae	forb	2	0.077	0.000	0.000	0.000	0.029
Sida spinosa	Malvaceae	forb	8	0.426	0.001	0.000	0.000	0.006
Sida trichopoda	Malvaceae	forb	33	0.093	0.338	0.182	1.378	0.268
Solanum esuriale	Solanaceae	forb	34	0.017	0.024	0.005	0.033	0.008
Sporobolus actinocladus	Poaceae	grass	1	0.000	0.000	0.000	0.000	0.118
Sporobolus caroli	Poaceae	grass	18	0.000	0.236	0.733	4.000	0.471
Sporobolus creber	Poaceae	grass	2	0.000	0.001	0.000	0.000	0.006
Sporobolus mitchellii	Poaceae	grass	25	0.233	0.749	0.683	0.013	0.039
Stemodia glabella	Scrophulariaceae	forb	1	0.000	0.001	0.000	0.000	0.000
Swainsona galegifolia	Fabaceae - Faboideae	forb	2	0.023	0.000	0.000	0.000	0.000
Swainsona greyana	Fabaceae - Faboideae	forb	3	0.000	0.000	0.000	0.000	0.002
Swainsona procumbens	Fabaceae - Faboideae	forb	4	0.000	0.000	0.035	0.005	0.000
Swainsona swainsoniodes	Fabaceae - Faboideae	forb	2	0.000	0.000	0.000	0.000	0.001
Teucrium racemosum	Lamiaceae	forb	4	0.000	0.001	0.000	0.000	0.002
Vachellia farnesiana	Fabaceae - Mimosoideae	shrub	10	0.195	0.001	0.000	0.000	0.412
Wahlenbergia communis	Campanulaceae	forb	20	0.002	0.005	0.002	0.000	0.302
Wahlenbergia gracilis	Campanulaceae	forb	2	0.001	0.000	0.000	0.000	0.001
Walwhalleya proluta	Poaceae	grass	1	0.000	0.001	0.000	0.000	0.000

Appendix 2: Pearson correlation matrix for MWD of aggregates and all other physico-chemical variables for nine surface (0–5 cm) soil samples.

	MWD	pH	EC	Ca	Mg	K	Na	ECEC	Ca:Mg	ECa	EMg	EK	ESP	TOC	N	C:N	Clay	Silt
pH	–0.653																	
EC	–0.253	0.335																
Ca	0.345	–0.281	–0.196															
Mg	0.164	–0.160	–0.410	0.873**														
K	–0.573	0.748*	0.726*	–0.182	–0.241													
Na	0.134	0.270	–0.547	0.335	0.297	–0.177												
ECEC	0.253	–0.176	–0.267	0.981**	0.943**	–0.154	0.374											
Ca:Mg	0.133	–0.206	0.449	–0.269	–0.692*	0.154	–0.175	–0.438										
ECa	0.429	–0.605	0.220	0.185	0.262	–0.203	–0.135	–0.005	0.844**									
EMg	0.006	–0.083	–0.540	0.437	0.814**	–0.338	0.095	0.575	–0.948**	–0.648								
EK	–0.522	0.641	0.660	–0.774*	–0.736*	0.744*	–0.391	–0.750*	0.274	–0.244	–0.504							
ESP	0.107	0.270	–0.538	0.083	0.053	–0.247	0.954**	0.112	–0.067	–0.113	–0.043	–0.265						
TOC	0.768*	–0.934**	–0.277	0.360	0.234	–0.652	–0.259	0.261	0.154	0.544	0.121	–0.603	–0.312					
N	0.620	–0.911**	–0.236	0.304	0.224	–0.549	–0.403	0.220	0.129	0.491	0.155	–0.516	–0.470	0.971**				
C:N	0.890**	–0.621	–0.277	0.346	0.141	–0.704*	0.270	0.244	0.163	0.472	–0.037	–0.599	0.297	0.658	0.458			
Clay	0.186	–0.407	–0.704	0.490	0.754*	–0.559	0.175	0.576	–0.717*	–0.333	0.848**	–0.705*	0.061	0.351	0.361	0.170		
Silt	0.148	–0.712*	–0.324	0.591	0.585	–0.560	–0.179	0.555	–0.166	0.315	0.443	–0.765*	–0.277	0.574	0.606	0.276	0.637	
Sand	–0.188	0.538	0.644	–0.562	–0.764*	0.607	–0.078	–0.618	0.603	0.156	–0.791*	0.783*	0.041	–0.450	–0.468	–0.217	–0.969**	–0.807**

*$P \leq 0.05$

**$P \leq 0.01$

Appendix 3: Pearson correlation matrix for MWD of aggregates and all other physico-chemical variables for 16 subsurface (20–30 cm) soil samples.

	MWD	pH	EC	Ca	Mg	K	Na	ECEC	Ca:Mg	ECa	EMg	EK	ESP	TOC	N	C:N	Clay	Silt
pH	–0.463																	
EC	–0.503	0.772																
Ca	–0.083	–0.137	–0.399															
Mg	–0.157	0.394	0.328	0.311														
K	–0.256	0.336	0.373	0.168	0.059													
Na	–0.420	0.807**	0.917**	–0.256	0.558*	0.332												
ECEC	–0.355	0.552*	0.457	0.442	0.871**	0.347	0.668**											
Ca:Mg	0.230	–0.534	–0.626	0.195	–0.842**	0.031	–0.733**	–0.650**										
ECa	0.310	–0.678	–0.830**	0.387	–0.664**	–0.187	–0.908**	–0.645**	0.887**									
EMg	0.061	0.128	0.129	0.019	0.798**	–0.310	0.273	0.408	–0.820**	–0.484								
EK	–0.046	0.030	0.117	–0.025	–0.367	0.881**	–0.004	–0.118	0.392	0.137	–0.559*							
ESP	–0.402	0.798	0.935**	–0.364	0.466	0.305	0.991**	0.565*	–0.698	–0.894**	0.230	0.016						
TOC	0.304	–0.736	–0.481	0.311	–0.229	–0.083	–0.529*	–0.185	0.426	0.465	–0.245	0.045	–0.556*					
N	0.404	–0.821	–0.634	0.295	–0.367	–0.206	–0.678**	–0.351	0.577	0.626**	–0.315	0.006	–0.692	0.959**				
C:N	–0.291	–0.268	0.136	0.050	0.222	0.297	0.027	0.191	–0.255	–0.197	0.206	0.200	–0.006	0.466	0.264			
Clay	–0.241	0.293	0.166	0.429	0.872**	0.130	0.433	0.869**	–0.613*	–0.522*	0.536*	–0.259	0.330	–0.144	–0.251	0.168		
Silt	0.043	–0.031	–0.103	0.280	0.009	0.086	–0.110	0.186	0.163	0.077	–0.233	0.032	–0.156	0.572*	0.511*	0.206	0.044	
Sand	0.238	–0.335	–0.186	–0.475	–0.716**	–0.194	–0.355	–0.813**	0.474	0.428	–0.341	0.181	–0.249	–0.157	–0.008	–0.289	–0.755	–0.637

*P < 0.05

**P < 0.01

Appendix 4: Spearman correlations between ASWAT and all physico-chemical variables describing surface soil (0–5 cm) samples from 60 sites.

	ASWAT	pH	EC	Ca	Mg	K	Na	ECEC	ECa	EMg	EK	ESP	Ca:Mg	TC	TOC	N	C:N	Clay	Silt
pH	0.050																		
EC	–0.250	0.199																	
Ca	–0.415**	–0.111	0.059																
Mg	0.019	0.054	–0.321*	0.537**															
K	0.010	0.062	0.282*	–0.073	–0.207														
Na	0.419**	0.481**	0.053	0.015	0.377**	–0.331*													
ECEC	–0.243	–0.038	–0.050	0.896**	0.827**	–0.103	0.229												
ECa	–0.413**	–0.319*	0.242	0.423**	–0.444**	0.141	–0.525**	0.026											
EMg	0.250	0.151	–0.433**	–0.076	0.742**	–0.307*	0.416**	0.299*	–0.857**										
EK	0.163	0.06	0.197	–0.559**	–0.618**	0.804**	–0.399**	–0.629**	0.092	–0.385**									
ESP	0.498**	0.498**	0.065	–0.219	0.161	–0.296*	0.958**	–0.024	–0.535**	0.329*	–0.225								
Ca:Mg	–0.319*	–0.235	0.367**	0.193	–0.659**	0.264*	–0.483**	–0.204	0.934**	–0.978**	0.303*	–0.427**							
TC	–0.234	–0.781**	–0.004	0.291*	0.126	–0.038	–0.396**	0.256*	0.295*	–0.089	–0.148	–0.477**	0.174						
TOC	–0.224	–0.818**	–0.030	0.261*	0.109	–0.016	–0.424**	0.229	0.291*	–0.085	–0.111	–0.493**	0.173	0.989**					
N	–0.193	–0.833**	–0.025	0.246	0.041	0.046	–0.463**	0.189	0.348**	–0.175	–0.037	–0.518**	0.250	0.962**	0.975**				
C:N	–0.258*	–0.030	0.118	0.287*	0.389**	–0.289*	0.155	0.378**	–0.083	0.270*	–0.470**	0.031	–0.211	0.379**	0.314*	0.155			
Clay	0.240	0.241	–0.444**	0.295*	0.756**	–0.285*	0.429**	0.518**	–0.439**	0.644**	–0.521**	0.292*	–0.598**	–0.130	–0.145	–0.171	0.158		
Silt	–0.328*	–0.265*	0.153	0.148	–0.024	0.132	–0.260*	0.067	0.200	–0.153	0.034	–0.285*	0.171	0.536**	0.508**	0.508**	0.111	–0.238	
Sand	–0.066	–0.101	0.365**	–0.267*	–0.715**	0.203	–0.258*	–0.475**	0.432**	–0.642**	0.430**	–0.118	0.596**	–0.147	–0.115	–0.088	–0.204	–0.782**	–0.342**

*P < 0.05

**P < 0.01

Appendix 5: Spearman correlations between ASWAT and all physico-chemical variables describing the subsurface soils (20–30 cm) at 60 sites.

	ASWAT	pH	EC	Ca	Mg	K	Na	ECEC	ECa	EMg	EK	ESP	Ca:Mg	TC	TOC	N	C:N	Clay	Silt
pH	0.486**																		
EC	0.476**	0.733**																	
Ca	–0.123	–0.222	–0.505**																
Mg	0.420**	0.098	0.076	0.301*															
K	0.381*8	0.313*	0.332*	0.184	0.255														
Na	0.715**	0.778**	0.848**	–0.249	0.341**	0.464**													
ECEC	0.324*	0.110	–0.067	0.744**	0.778**	0.431**	0.273*												
ECa	–0.616**	–0.441**	–0.606**	0.290*	–0.706**	–0.368**	–0.717**	–0.374**											
EMg	0.262*	–0.005	0.097	–0.142	0.817**	–0.040	0.183	0.337*	–0.732**										
EK	0.237	0.278*	0.370**	–0.076	–0.045	0.915**	0.371**	0.085	–0.221	–0.205									
ESP	0.656**	0.795**	0.917**	–0.439**	0.188	0.354**	0.965**	0.068	–0.672**	0.139	0.337*								
Ca:Mg	–0.432**	–0.185	–0.325*	0.214*	–0.817**	–0.124	–0.422**	–0.379**	0.901**	–0.944**	0.039	–0.378**							
TC	–0.414**	–0.656**	–0.376**	–0.036	–0.102	–0.263*	–0.482**	–0.213	0.222	–0.014	–0.184	–0.470**	0.110						
TOC	–0.416**	–0.676**	–0.404**	0.001	–0.093	–0.282*	–0.490**	–0.197	0.246	–0.023	–0.219	–0.493**	0.128	0.986**					
N	–0.463**	–0.712**	–0.454**	0.064	–0.143	–0.287*	–0.526**	–0.159	0.329*	–0.111	–0.214	–0.521**	0.216	0.936**	0.945**				
C:N	0.018	–0.075	0.144	–0.273*	0.098	–0.031	0.022	–0.150	–0.264*	0.256	–0.012	0.055	–0.280*	0.487**	0.423**	0.197			
Clay	0.435**	0.212	0.061	0.508**	0.819**	0.429**	0.373**	0.858**	–0.504**	0.501**	0.134	0.201	–0.538**	–0.323*	–0.321*	–0.317*	–0.039		
Silt	–0.282*	–0.224	–0.165	0.015	–0.153	–0.281*	–0.209	–0.153	0.222	–0.132	–0.234	–0.200	0.194	0.590**	0.579**	0.570**	0.174	–0.228	
Sand	–0.182	–0.078	0.072	–0.396**	–0.605**	–0.193	–0.175	–0.610**	0.295*	–0.365**	0.023	–0.033	0.348**	–0.123	–0.112	–0.084	–0.118	–0.686**	–0.498**

*P ≤ 0.05

**P ≤ 0.01

Appendix 6: Spearman rank correlations between native plant species richness ranking, maximum rarity ranking, NMDS axes, the five principal components and individual environmental variables.

	Nat. rich.	Max. rare	PC1	PC2	PC3	PC4	PC5	NMDS1	NMDS2	pH	EC	Ca	Mg	K	Na	ECEC	EK1	EK2	Clay	Silt	Sand	N	Flooding
Max. rare	0.28*																						
PC1	0.29*	–0.16																					
PC2	0.08	0.42**	0.06																				
PC3	0.03	–0.15	0.02	–0.07																			
PC4	–0.08	0.05	0.07	0.08	0.07																		
PC5	–0.02	–0.43**	0.17	–0.01	–0.04	0.06																	
NMDS1	–0.15	0.27*	–0.73**	0.15	0.09	–0.29*	–0.36*																
NMDS2	–0.25	0.00	–0.10	0.44**	0.05	0.20	0.04	–0.04															
pH	0.21	–0.40**	0.72**	–0.42**	–0.02	0.18	0.24	–0.73**	–0.18														
EC	0.21	–0.25	0.81**	–0.27*	–0.01	0.16	–0.06	–0.71**	–0.22	0.73**													
Ca	–0.28*	–0.26	–0.68**	–0.51**	0.01	0.01	0.25	0.25	–0.12	–0.19	–0.45**												
Mg	–0.12	–0.05	–0.47**	–0.77**	–0.14	–0.20	–0.28*	0.28*	–0.39**	–0.08	–0.10	0.54**											
K	0.14	–0.27*	0.22	–0.51**	0.81**	–0.04	–0.06	–0.19	–0.25	0.33*	0.35*	0.06	0.23										
Na	0.08	–0.27*	0.55**	–0.56**	–0.11	0.13	–0.06	–0.55**	–0.33*	0.67**	0.79**	–0.14	0.22	0.32*									
ECEC	–0.20	–0.27*	–0.45**	–0.87**	–0.03	–0.04	–0.04	0.13	–0.34*	0.09	–0.07	0.78**	0.88**	0.30*	0.31*								
EK1	0.03	–0.17	0.41**	0.22	0.11	0.54**	0.61**	–0.59**	0.26	0.25	0.31*	–0.22	–0.46**	0.05	0.20	–0.31*							
EK2	0.20	–0.18	0.39**	–0.22	0.86**	0.01	–0.03	–0.22	–0.13	0.29*	0.38*	–0.21	–0.06	0.93**	0.24	–0.02	0.22						
Clay	–0.14	–0.28*	–0.37*	–0.89*	0.04	–0.07	–0.03	0.04	–0.30*	0.11	–0.03	0.64**	0.82**	0.36*	0.28*	0.88**	–0.28*	0.08					
Silt	–0.11	0.23	–0.39**	0.29*	0.21	0.66**	–0.25	0.25	0.40**	–0.25	–0.28*	0.12	–0.17	–0.13	–0.35**	–0.08	0.05	–0.10	–0.15				
Sand	0.23	0.08	0.57**	0.55**	–0.21	–0.41**	0.20	–0.24	0.01	0.06	0.24	–0.62**	–0.56**	–0.23	0.00	–0.68**	0.23	–0.01	–0.72**	–0.52**			
N	–0.18	0.25	–0.61**	0.42**	0.12	0.26	–0.31*	0.54**	0.32*	–0.69**	–0.52**	0.17	–0.03	–0.20	–0.49**	–0.06	–0.03	–0.13	–0.19	0.59**	–0.21		
Flooding	–0.09	0.10	–0.44**	0.09	0.01	–0.58**	0.01	0.54**	0.05	–0.43**	–0.46**	0.12	0.16	–0.17	–0.36*	0.04	–0.32*	–0.18	0.03	–0.06	0.04	0.11	
Mean rain	–0.22	0.28*	–0.73**	0.29*	–0.16	–0.27*	–0.25	0.65**	0.24	–0.72**	–0.64**	0.28*	0.17	–0.38**	–0.50**	0.07	–0.32*	–0.41**	–0.02	0.18	–0.10	0.60**	0.45**

Codes: Nat. rich. = native species richness; Max. rare = maximum rarity; PC1–PC5 = PCs from Table 6.4; NMDS1, 2 = coordinates from 2-D NMDS axes; Flooding = number of years since the site was last flooded.

Appendix 7: Spearman rank correlations between native bird species richness ranking, maximum rarity ranking, NMDS axes, the five principal components and individual habitat variables.

	Max. rare	PC1	PC2	PC3	PC4	PC5	NMDS1	NMDS2	Canopycov	tshrubcov	lshrubcov	herbcover	Litcover	Totground	Woodbio	Herbbio	litbio	CWDbio	Noltrees	Nostrees	Nosap	Noshrub	Nosnags	NoCWD	TreeSppRi	ShrubSppR
PC1	–0.34*																									
PC2	–0.24	–0.02																								
PC3	–0.18	–0.03	–0.09																							
PC4	–0.24	–0.01	0.05	–0.08																						
PC5	–0.04	0.02	0.14	–0.12	0.08																					
NMDS1	0.30*	–0.17	–0.54**	–0.07	0.00	–0.04																				
NMDS2	–0.39**	0.55**	–0.19	–0.02	0.23	0.15	0.05																			
Canopycov	0.24	–0.90**	0.03	0.23	0.04	0.06	0.09	–0.43**																		
tshrubcov	0.26	–0.83**	0.19	0.26	–0.27*	–0.07	–0.03	–0.59**	0.78**																	
lshrubcov	–0.17	0.10	0.27*	0.12	0.17	–0.64**	–0.19	–0.13	–0.11	0.00																
herbcover	0.03	0.42**	–0.68**	0.46**	–0.07	–0.17	0.25	0.33*	–0.35*	–0.39**	–0.12															
Litcover	0.40**	–0.58**	–0.60**	–0.13	–0.04	–0.31*	0.55**	–0.23	0.46**	0.32*	–0.08	0.09														
Totground	0.37*	–0.31*	–0.87**	0.15	–0.11	–0.29*	0.54**	–0.06	0.27*	0.13	–0.17	0.57**	0.83**													
Woodbio	0.31*	–0.81**	–0.29*	0.24	0.28*	–0.05	0.29*	–0.36*	0.81**	0.60**	–0.18	–0.04	0.61**	0.52**												
Herbbio	–0.21	0.46**	–0.33*	0.63**	–0.21	–0.21	0.04	0.23	–0.34*	–0.26	0.01	0.69**	–0.22	0.16	–0.19											
litbio	0.34*	–0.67**	–0.49**	–0.18	–0.01	–0.15	0.45**	–0.35*	0.52**	0.40**	–0.20	–0.06	0.86**	0.66**	0.65**	–0.25										
CWDbio	0.29*	–0.82*	–0.09	–0.07	0.27*	–0.08	0.24	–0.43**	0.70**	0.59**	–0.12	–0.28*	0.49**	0.31*	0.79**	–0.38*	0.57**									
Noltrees	0.14	–0.58**	–0.39**	0.31*	0.43**	–0.05	0.29*	–0.06	0.62**	0.40**	–0.11	0.06	0.51**	0.48**	0.83**	–0.04	0.51**	0.61**								
Nostrees	0.21	–0.87**	0.27*	0.02	–0.14	0.01	–0.07	–0.57**	0.80**	0.78**	0.00	–0.51**	0.34**	0.05	0.58**	–0.41**	0.45**	0.65**	0.32**							
Nosap	0.28*	–0.79**	0.15	0.17	–0.34*	–0.01	0.08	–0.53**	0.75**	0.83**	–0.04	–0.39**	0.35*	0.12	0.55**	–0.28*	0.45**	0.51**	0.30*	0.79**						
Noshrub	0.00	–0.45**	0.46**	0.38*	–0.24	–0.35**	–0.20	–0.48**	0.42**	0.64**	0.26	–0.34*	0.05	–0.13	0.23	–0.03	0.05	0.30*	0.11	0.49**	0.48**					
Nosnags	0.21	–0.79**	0.12	–0.24	0.05	–0.02	–0.01	–0.48**	0.63**	0.60**	–0.10	–0.41**	0.36*	0.13	0.57**	–0.45**	0.51**	0.74**	0.35*	0.77**	0.58**	0.26				
NoCWD	0.14	–0.80**	–0.05	–0.01	0.28*	–0.04	0.15	–0.28*	0.72**	0.58**	–0.14	–0.33*	0.39**	0.22	0.75**	–0.30*	0.53**	0.83**	0.61**	0.65**	0.49**	0.28*	0.67**			
TreeSppRi	–0.11	–0.47**	0.14	0.30*	0.32*	0.32*	0.02	–0.20	0.50**	0.39**	0.04	–0.20	0.12	0.01	0.41**	–0.26	0.25	0.30*	0.39**	0.36*	0.39**	0.22	0.28*	0.29*		
ShrubSppR	0.03	–0.47**	0.50**	0.02	0.23	–0.08	–0.26	–0.39**	0.37*	0.41**	0.12	–0.39**	–0.05	–0.22	0.28*	–0.32*	0.05	0.34*	0.10	0.42**	0.30*	0.48**	0.51**	0.33*	0.30*	
natrichra	0.68**	–0.69**	–0.15	0.06	–0.11	–0.11	0.25	–0.59**	0.62**	0.59**	–0.12	–0.14	0.49**	0.39**	0.68**	–0.30*	0.52**	0.59**	0.43**	0.51**	0.57**	0.33*	0.45**	0.47**	0.18	0.27*

Codes: Max. rare = maximum rarity; PC1–PC5 = PCs from Table 6.4; NMDS1, 2 = coordinates from 2-D NMDS axes; Canopycov = foliage projective cover of the tree layer; tshrubcov = foliage projective cover of the tall shrub layer lshrubcov = foliage projective cover of the low shrub layer; herbcov = foliage projective cover of herbaceous vegetation; litcov = % cover of dead, unattached plant material; Totground = total groundcover; Woodbio = woody biomass; Herbbio = herbaceous biomass; litbio = litter biomass; CWDbio = CWD biomass; Noltrees = number of trees >50 cm DBH; Nostrees = number of trees 10 – 50 cm DBH; Nosap = number of saplings (woody recruits < 10 cm); Noshrub = number of shrubs < 2 m high; Nosnags = number of standing dead trees; NoCWD = number of pieces of CWD; TreeSppRi = number of tree species; ShrubSppR = number of tall shrub species

Appendix 8: Average abundance and frequency of occurrence (%) of vascular plant species in five vegetation types common to the lower Namoi floodplain

	Vegetation type									
	River red gum		Coolibah		Grassland		Myall		Black box	
	Average abundance	Proportion of sites	Average abundance	Proportion of sites	Average abundance	Proportion of sites	Average abundance	Proportion of sites	Average abundance	Proportion of sites
Abutilon oxycarpum	0.00	0.00	0.02	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Acacia pendula	0.00	0.00	0.29	5.88	0.00	6.67	17.17	100.00	0.00	0.00
Acacia salicina	3.33	33.33	0.12	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Acacia stenophylla	4.17	50.00	1.94	47.06	0.00	0.00	0.00	0.00	0.00	0.00
Alectryon oleofolius	0.00	8.33	0.53	47.06	0.01	33.33	0.00	33.33	0.01	50.00
Alternanthera denticulata	0.01	33.33	0.05	52.94	0.03	20.00	0.00	16.67	0.01	75.00
Alternanthera nodiflora	0.01	25.00	0.00	23.53	0.00	6.67	0.00	0.00	0.00	0.00
Alternanthera pungens	0.04	25.00	0.00	0.00	0.03	6.67	0.00	0.00	0.00	0.00
Alyssum linifolium	0.04	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Amaranthus albus	0.00	16.67	0.05	35.29	0.04	20.00	0.00	0.00	0.13	25.00
Amaranthus hybridus	0.04	33.33	0.00	5.88	0.00	6.67	0.00	0.00	0.00	0.00
Amaranthus powellii	0.00	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Ammi majus	0.58	33.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Amyema linophyllum	0.00	0.00	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Amyema quandang	0.00	0.00	0.00	0.00	0.00	0.00	0.08	16.67	0.00	0.00
Apophyllum anomalum	0.00	0.00	0.00	0.00	0.00	13.33	0.17	33.33	0.00	25.00
Arctotheca calendula	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Argemone ochroleuca	0.02	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Aristida calycina	0.00	0.00	0.00	0.00	0.00	0.00	0.00	16.67	0.00	0.00
Aristida leptopoda	0.00	0.00	0.00	0.00	0.03	6.67	0.00	0.00	0.00	25.00
Aristida ramosa	0.00	8.33	0.00	11.76	0.00	0.00	0.00	0.00	0.00	0.00
Asperula conferta	1.03	41.67	0.27	52.94	0.00	0.00	0.00	16.67	0.00	0.00
Aster subulatus	0.00	8.33	0.00	0.00	0.00	13.33	0.00	16.67	0.00	25.00
Astrebla lappacea	0.00	0.00	0.02	5.88	7.33	33.33	0.58	50.00	0.33	75.00
Atalaya hemiglauca	0.00	0.00	0.00	5.88	0.00	6.67	0.00	0.00	0.00	25.00
Atriplex leptocarpa	0.00	0.00	0.09	5.88	0.00	13.33	2.19	50.00	0.77	100.00

Atriplex lindleyi	0.00	0.00	0.06	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Atriplex muelleri	0.00	8.33	0.03	11.76	0.05	13.33	0.22	50.00	1.88	75.00
Atriplex semibaccata	0.04	8.33	0.38	35.29	0.51	46.67	0.96	100.00	1.38	75.00
Austrostipa aristiglumis	0.26	25.00	0.03	11.76	2.23	33.33	0.08	16.67	0.00	0.00
Austrodanthonia bipartita	0.00	0.00	0.55	23.53	0.37	13.33	0.00	16.67	0.00	0.00
Austrostipa scabra subsp. scabra	0.00	0.00	0.01	5.88	0.00	13.33	0.07	16.67	0.00	0.00
Austrostipa verticillata	0.00	16.67	0.00	0.00	0.00	0.00	0.00	16.67	0.00	0.00
Avena barbata	0.00	8.33	0.02	11.76	0.03	26.67	0.00	0.00	0.00	0.00
Bidens pilosa	0.16	41.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Bidens subalternans	0.53	58.33	0.00	5.88	0.00	0.00	0.00	16.67	0.00	0.00
Boerhavia dominii	0.36	83.33	0.51	94.12	0.41	93.33	0.54	66.67	0.67	100.00
Bothriochloa bladhii	0.01	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Bothriochloa macra	0.09	33.33	0.00	0.00	0.00	13.33	0.00	0.00	0.00	0.00
Brachyscome curvicarpa	0.00	8.33	0.00	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Brachyscome dentata	0.00	0.00	0.00	11.76	0.00	0.00	0.00	0.00	0.00	0.00
Bromus cartharticus	1.14	75.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Bromus molliformis	0.00	0.00	0.00	0.00	0.03	6.67	0.00	0.00	0.00	0.00
Bulbine semibarbata	0.02	16.67	0.08	52.94	0.02	20.00	0.00	16.67	0.00	75.00
Calendula arvensis	0.00	0.00	0.15	5.88	0.10	26.67	0.06	50.00	1.25	75.00
Calotis erinacea	0.00	8.33	0.00	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Calostemma purpureum	0.10	25.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Calotis scabiosifolia var. scabiosifolia	0.00	0.00	0.00	11.76	0.00	20.00	0.00	33.33	0.01	75.00
Calotis scapigera	0.00	8.33	0.08	29.41	0.00	0.00	0.00	0.00	0.00	0.00
Capparis lasiantha	0.00	0.00	0.06	23.53	0.00	6.67	0.00	0.00	0.00	0.00
Capparis mitchellii	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Carex inversa	1.30	83.33	0.10	17.65	0.09	46.67	0.00	0.00	0.00	25.00
Carthamus lanatus	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Casuarina cristata	0.00	0.00	0.65	29.41	0.00	0.00	0.00	16.67	0.00	0.00
Cenchrus ciliaris	0.00	0.00	0.00	0.00	0.13	6.67	0.00	0.00	0.00	0.00
Centaurea melitensis	0.00	0.00	0.00	0.00	0.07	33.33	0.00	0.00	0.00	0.00
Chamaesyce drummondii	0.17	16.67	0.01	47.06	0.00	13.33	0.01	83.33	0.01	100.00

Chenopodium album	0.06	25.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Chloris truncata	0.04	16.67	0.21	17.65	0.11	33.33	0.08	16.67	0.06	25.00
Chloris ventricosa	0.04	8.33	0.06	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Cichorium intybus	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Cirsium vulgare	0.01	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Commelina cyanea	0.21	33.33	0.00	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Convolvulus arvensis	0.00	16.67	0.00	29.41	0.02	46.67	0.00	16.67	0.00	0.00
Convolvulus erubescens	0.00	8.33	0.00	0.00	0.02	6.67	0.00	0.00	0.00	0.00
Conyza bonariensis	0.00	50.00	0.00	0.00	0.07	20.00	0.01	16.67	0.00	0.00
Crassula sieberiana	0.00	0.00	0.00	0.00	0.00	0.00	0.00	16.67	0.00	0.00
Craspedia variabilis	0.00	16.67	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Cucumis melo	0.00	16.67	0.06	47.06	0.00	33.33	0.04	16.67	0.00	0.00
Cucumis myriocarpus subsp. leptodermis	0.06	41.67	0.00	11.76	0.06	53.33	0.08	16.67	0.00	25.00
Cullen tenax	0.00	8.33	0.00	0.00	0.00	26.67	0.00	0.00	0.00	0.00
Cyclospermum leptophyllum	0.01	58.33	0.00	0.00	0.04	26.67	0.00	16.67	0.00	0.00
Cynodon dactylon	0.75	66.67	0.00	5.88	0.00	6.67	0.00	0.00	0.00	0.00
Cyperus bifax	0.09	16.67	0.03	52.94	0.38	26.67	0.01	16.67	0.00	0.00
Cyperus difformis	0.00	0.00	0.06	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Cyperus gracilis	0.00	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Cyperus lucidus	0.67	33.33	0.15	52.94	0.15	33.33	0.00	0.00	0.00	25.00
Dactyloctenium radulans	0.00	0.00	0.00	0.00	0.00	13.33	0.00	0.00	0.00	25.00
Datura ferox	0.00	25.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Daucus carota	0.00	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Daucus glochidiatus	0.00	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Dichanthium sericeum subsp. sericeum	0.83	16.67	0.03	5.88	0.73	33.33	0.00	0.00	0.00	0.00
Dichondra sp. A	0.04	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Digitaria coenicola	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Digitaria divaricatissima	0.00	0.00	0.00	0.00	0.13	6.67	0.00	16.67	0.06	25.00
Dissocarpus biflorus var. cephalocarpus	0.00	0.00	0.59	5.88	0.13	6.67	1.27	33.33	0.56	75.00
Echinochloa colona	0.47	66.67	0.00	11.76	0.00	20.00	0.00	0.00	0.00	0.00

Echium plantagineum	0.06	25.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eclipta platyglossa	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Einadia hastata	0.00	8.33	0.15	5.88	0.00	6.67	0.83	16.67	0.00	0.00
Einadia nutans	2.03	83.33	3.72	100.00	0.90	66.67	2.25	100.00	2.19	100.00
Eleocharis plana	0.63	25.00	0.03	11.76	0.00	6.67	0.00	0.00	0.00	0.00
Elymus plurinervis	0.00	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Enchylaena tomentosa	0.00	0.00	0.00	17.65	0.00	6.67	0.18	33.33	0.01	25.00
Enteropogon acicularis	0.63	58.33	3.27	76.47	5.20	80.00	2.63	100.00	0.31	50.00
Eragrostis cilianensis	0.34	25.00	0.01	11.76	0.37	40.00	0.00	16.67	0.00	0.00
Eragrostis parviflora	0.00	0.00	0.00	0.00	0.00	20.00	0.00	0.00	0.00	0.00
Eragrostis setifolia	0.00	0.00	0.32	17.65	0.00	0.00	0.02	50.00	0.13	25.00
Eremophila bignonifolia	0.17	8.33	0.41	23.53	0.00	0.00	0.01	16.67	3.00	75.00
Eremophila debilis	0.48	25.00	0.12	29.41	0.00	6.67	0.00	16.67	0.00	25.00
Eremophila maculata	0.00	0.00	0.00	11.76	0.00	6.67	0.33	33.33	0.24	50.00
Eremophila mitchellii	0.00	0.00	0.00	0.00	0.00	0.00	0.17	16.67	0.25	25.00
Eriochloa crebra	0.01	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Eriochloa procera	0.00	0.00	0.89	17.65	0.00	6.67	0.08	16.67	0.00	0.00
Eriochloa pseudoacrotricha	0.77	75.00	0.52	70.59	2.00	46.67	0.51	33.33	0.00	25.00
Erodium crinitum	0.00	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	75.00
Eryngium paludosum	0.00	0.00	0.03	5.88	0.00	6.67	0.00	0.00	0.00	0.00
Eucalyptus camaldulensis	12.67	100.00	0.88	11.76	0.00	6.67	0.00	0.00	0.00	0.00
Eucalyptus coolabah	3.17	33.33	14.29	100.00	0.47	40.00	0.00	16.67	1.50	50.00
Eucalyptus largiflorens	0.00	0.00	0.00	0.00	0.01	6.67	2.00	33.33	10.50	100.00
Eucalyptus populnea	0.00	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Euchiton sphaericus	0.02	16.67	0.00	17.65	0.02	33.33	0.04	16.67	0.00	50.00
Euphorbia stevenii	0.00	8.33	0.06	23.53	0.04	33.33	0.00	0.00	0.00	0.00
Fumaria capreolata subsp. capreolata	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Geranium solanderi var. solanderi	0.00	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Glycine tabacina	0.01	41.67	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Gomphocarpus fruticosus	0.06	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Goodenia fascicularis	0.00	25.00	0.28	82.35	0.00	26.67	0.05	50.00	0.13	75.00

Goodenia glauca	0.00	8.33	0.02	23.53	0.00	0.00	0.00	16.67	0.00	0.00
Haloragis glauca f. glauca	0.00	16.67	0.01	29.41	0.00	6.67	0.00	16.67	0.00	0.00
Harmsiodoxa brevipes var major	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Hibiscus brachysiphonius	0.00	0.00	0.00	5.88	0.00	6.67	0.00	16.67	0.10	50.00
Hibiscus trionum	0.09	16.67	0.08	35.29	0.12	26.67	0.00	0.00	0.00	0.00
Hirschfeldia incana	0.00	0.00	0.38	17.65	0.50	6.67	0.17	16.67	0.00	25.00
Hordeum vulgare	0.00	0.00	0.00	5.88	0.83	13.33	0.04	33.33	0.00	0.00
Hypochaeris microcephala var. albiflora	0.00	25.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Ipomoea lonchophylla	0.00	0.00	0.02	23.53	0.52	40.00	0.00	16.67	0.00	0.00
Iseilema vaginiflorum	0.00	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Jasminum suavissimum	0.02	25.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Juncus aridicola	0.00	8.33	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Lactuca saligna	0.13	50.00	0.00	0.00	0.00	20.00	0.00	16.67	0.00	0.00
Lamium amplexicaule	0.01	25.00	0.00	5.88	0.00	6.67	0.00	0.00	0.00	0.00
Leiocarpa brevicompta	0.00	0.00	0.07	11.76	0.00	0.00	0.00	0.00	0.00	0.00
Leiocarpa leptolepis	0.00	0.00	0.00	0.00	0.03	13.33	0.00	0.00	0.00	0.00
Leiocarpa tomentosa	0.00	0.00	0.02	11.76	0.00	6.67	0.05	16.67	0.00	0.00
Lepidium africanum	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Lepidium bonariense	0.05	41.67	0.00	11.76	0.00	6.67	0.00	0.00	0.00	0.00
Lepidium fasciculatum	0.13	25.00	0.00	11.76	0.03	13.33	0.00	33.33	0.00	25.00
Lepidium muelleri-ferdinandi	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Lepidium pseudohyssopifolium	0.04	25.00	0.00	0.00	0.38	33.33	0.00	16.67	0.00	25.00
Leptochloa ciliolata	0.21	16.67	0.00	0.00	0.00	13.33	0.00	0.00	0.00	0.00
Leptochloa digitata	0.01	16.67	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Leptochloa divaricatissima	0.01	41.67	0.83	64.71	0.84	26.67	0.00	16.67	0.00	0.00
Lolium loliaceum	0.23	25.00	0.00	0.00	0.17	6.67	0.00	0.00	0.00	0.00
Lolium perenne	2.74	41.67	0.00	0.00	0.07	6.67	0.00	0.00	0.00	0.00
Lomandra longifolia	0.67	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Lycium ferrocisimum	0.00	0.00	0.06	5.88	0.00	0.00	0.18	33.33	0.00	25.00
Maireana aphylla	0.00	0.00	0.12	5.88	0.00	0.00	0.10	16.67	0.00	0.00

Maireana coronata	0.00	0.00	0.00	0.00	0.13	6.67	0.01	16.67	0.00	25.00
Maireana decalvans	0.03	8.33	0.06	23.53	0.03	6.67	0.18	33.33	0.06	25.00
Malvastrum americanum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	16.67	0.00	0.00
Malvastrum coromandelianum	0.00	8.33	0.03	23.53	0.04	26.67	0.04	33.33	0.19	50.00
Malva parviflora	0.13	25.00	0.00	5.88	0.00	6.67	0.04	16.67	0.00	0.00
Marrubium vulgare	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Marsilea drummondii	0.17	58.33	0.11	52.94	0.05	33.33	0.01	33.33	0.13	25.00
Medicago minima	0.00	0.00	0.00	5.88	0.17	20.00	0.01	33.33	0.01	25.00
Medicago polymorpha	2.99	83.33	0.28	70.59	1.88	66.67	0.04	33.33	0.00	25.00
Medicago sativa	0.00	0.00	0.00	0.00	0.67	6.67	0.00	0.00	0.00	0.00
Melaleuca bracteata	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Melia azedarach	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Muehlenbeckia florulenta	1.22	41.67	2.24	41.18	0.03	6.67	0.09	33.33	1.00	75.00
Myoporum montanum	0.00	25.00	0.00	5.88	0.00	6.67	0.00	16.67	0.00	0.00
Neptunia gracilis	0.00	0.00	0.05	23.53	0.02	53.33	0.00	0.00	0.00	0.00
Nicotiana simulans	0.00	8.33	0.00	29.41	0.02	13.33	0.17	33.33	0.00	0.00
Opuntia aurantiaca	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Opuntia stricta	0.00	8.33	0.00	5.88	0.00	6.67	0.00	0.00	0.07	50.00
Oxalis corniculata	0.00	8.33	0.00	0.00	0.00	20.00	0.00	0.00	0.00	0.00
Oxalis perennans	0.03	66.67	0.05	64.71	0.08	73.33	0.01	33.33	0.00	0.00
Panicum buncei	0.00	0.00	0.00	5.88	0.00	13.33	0.00	0.00	0.00	0.00
Panicum coloratum	0.13	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Panicum decompositum	1.54	33.33	0.36	52.94	3.53	86.67	0.38	33.33	0.14	25.00
Panicum queenslandicum var. queenslandicum	0.00	0.00	0.04	5.88	1.42	13.33	0.00	0.00	0.00	0.00
Paspalum dilatatum	0.21	25.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Paspalum distichum	0.17	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Paspalidium globoideum	0.00	16.67	0.00	0.00	0.03	13.33	0.00	0.00	0.00	0.00
Paspalidium gracile	2.56	75.00	0.54	58.82	1.35	46.67	2.38	83.33	3.25	100.00
Paspalidium jubiflorum	9.31	100.00	3.98	94.12	1.28	40.00	0.33	33.33	0.00	0.00
Persicaria prostrata	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Phalaris paradoxa	1.17	33.33	0.00	11.76	0.10	26.67	0.00	0.00	0.00	0.00

Phyla canescens	19.17	75.00	0.02	11.76	0.50	20.00	0.00	0.00	0.00	0.00
Phyllanthus virgatus	0.00	8.33	0.04	35.29	0.00	0.00	0.00	0.00	0.00	0.00
Physalis lanceifolia	0.03	50.00	0.00	29.41	0.05	33.33	0.00	0.00	0.00	0.00
Plantago lanceolata	0.00	0.00	0.00	5.88	0.00	0.00	0.00	16.67	0.00	0.00
Plantago turrifera	0.00	0.00	0.00	11.76	0.00	20.00	0.00	16.67	0.00	25.00
Poa labillardierei var. labillardierei	0.02	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Polygonum aviculare	0.00	16.67	0.00	0.00	0.01	13.33	0.00	0.00	0.00	0.00
Polymeria longifolia	0.00	25.00	0.00	5.88	0.37	33.33	0.21	33.33	0.22	75.00
Portulaca oleracea	0.03	25.00	0.17	64.71	0.50	53.33	0.23	100.00	0.44	100.00
Ptilotus semilanatus	0.00	0.00	0.02	5.88	0.07	20.00	0.13	33.33	0.14	75.00
Pycnosorus globosus	0.00	0.00	0.00	0.00	0.00	20.00	0.00	33.33	0.00	25.00
Ranunculus lappaceus	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Rapistrum rugosum	1.86	100.00	2.13	94.12	3.45	100.00	2.30	100.00	1.26	100.00
Rhagodia spinescens	0.00	8.33	0.06	5.88	0.00	0.00	0.17	16.67	0.00	0.00
Rhodanthe floribunda	0.00	0.00	0.00	0.00	0.00	13.33	0.00	0.00	0.00	0.00
Rhynchosia australis	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Rhynchosia minima	0.00	0.00	0.00	5.88	0.10	6.67	0.01	16.67	0.00	0.00
Rostellularia adscendens var. adscendens	0.00	0.00	0.00	5.88	0.02	6.67	0.02	16.67	0.25	75.00
Rumex brownii	0.74	91.67	0.00	23.53	0.09	20.00	0.00	0.00	0.00	0.00
Salsola kali	0.42	50.00	0.78	64.71	0.46	80.00	1.88	83.33	5.56	100.00
Salvia reflexa	0.00	8.33	0.00	0.00	0.00	0.00	0.04	16.67	0.00	0.00
Scleroblitum atriplicinum	0.00	8.33	0.00	5.88	0.00	0.00	0.00	33.33	0.00	25.00
Sclerolaena bicornis var. horrida	0.00	0.00	0.44	5.88	1.10	13.33	1.92	50.00	2.44	100.00
Sclerolaena birchii	0.29	25.00	0.35	11.76	0.17	26.67	0.01	16.67	0.00	25.00
Sclerolaena calcarata	0.00	0.00	0.09	17.65	0.22	13.33	0.85	66.67	2.06	100.00
Sclerolaena muricata	1.50	50.00	6.93	100.00	5.28	86.67	5.42	100.00	2.39	100.00
Sclerolaena stelligera	0.00	0.00	0.15	5.88	0.07	13.33	0.00	16.67	0.15	75.00
Senecio glossanthus	0.00	0.00	0.01	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Senecio quadridentatus	0.00	0.00	0.00	11.76	0.00	0.00	0.00	0.00	0.00	0.00
Senna barclayana	0.11	25.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00

Sesbania cannabina	0.00	16.67	0.00	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Sida ammophila	0.00	8.33	0.00	11.76	0.00	6.67	0.00	0.00	0.00	50.00
Sida corrugata	0.04	8.33	0.00	5.88	0.02	13.33	0.00	0.00	0.00	0.00
Sida rhombifolia	0.88	66.67	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Sida spinosa	1.03	66.67	0.00	5.88	0.04	20.00	0.00	0.00	0.00	0.00
Sida trichopoda	0.05	33.33	0.23	58.82	0.41	80.00	0.18	83.33	0.90	100.00
Sigesbeckia orientalis subsp. orientalis	0.13	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Silybum marianum	0.06	25.00	0.00	0.00	0.03	20.00	0.00	0.00	0.00	0.00
Sinapis arvensis	0.00	0.00	0.00	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Sisymbrium irio	0.00	0.00	0.06	17.65	0.10	26.67	0.13	66.67	0.00	50.00
Solanum esuriale	0.03	25.00	0.03	76.47	0.08	86.67	0.05	83.33	0.03	100.00
Solanum nigrum	0.07	66.67	0.02	11.76	0.00	0.00	0.00	16.67	0.00	0.00
Sonchus asper	0.00	16.67	0.00	17.65	0.00	20.00	0.00	0.00	0.00	0.00
Sonchus oleraceus	0.05	83.33	0.02	76.47	0.26	86.67	0.01	100.00	0.00	75.00
Sorghum bicolor	0.00	8.33	0.00	0.00	0.07	6.67	0.00	0.00	0.00	0.00
Sporobolus actinocladus	0.00	0.00	0.00	0.00	0.20	6.67	0.00	0.00	0.00	0.00
Sporobolus caroli	0.00	25.00	0.49	47.06	0.65	66.67	0.78	83.33	3.63	100.00
Sporobolus creber	0.00	0.00	0.00	5.88	0.02	13.33	0.00	16.67	0.00	0.00
Sporobolus mitchellii	0.38	50.00	0.73	82.35	0.05	26.67	0.59	50.00	0.32	50.00
Stachys arvensis	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stellaria media	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Stemodia glabella	0.00	0.00	0.00	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Swainsona galegifolia	0.01	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Swainsona greyana	0.00	0.00	0.00	0.00	0.00	13.33	0.00	0.00	0.00	0.00
Swainsona procumbens	0.00	0.00	0.00	0.00	0.00	0.00	0.03	33.33	0.00	50.00
Swainsona swainsoniodes	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Tetragonia tetragonioides	0.75	41.67	0.65	52.94	0.00	0.00	0.42	33.33	0.00	25.00
Teucrium racemosum	0.00	0.00	0.00	5.88	0.00	20.00	0.00	0.00	0.00	0.00
Trianthema triquetra	0.48	25.00	0.00	23.53	0.00	26.67	0.34	50.00	0.02	50.00
Tribulus micrococcus	0.17	8.33	0.09	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Tribulus terrestris	2.38	66.67	0.82	88.24	1.35	93.33	0.80	100.00	0.90	100
Trifolium glomeratum	0.00	0.00	0.00	0.00	0.03	6.67	0.00	0.00	0.00	0.00

Tripogon loliiformis	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Urochloa panicoides	1.57	50.00	0.15	11.76	1.90	33.33	0.00	0.00	0.00	0.00
Urtica urens	0.01	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Vachellia farnisiana	0.21	50.00	0.00	5.88	0.47	13.33	0.00	0.00	0.00	0.00
Verbena bonariensis	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Verbena officinalis	0.00	16.67	0.01	35.29	0.00	46.67	0.00	33.33	0.00	0.00
Vicia sativa	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Vittadinia cuneata	0.00	8.33	0.00	5.88	0.02	13.33	0.00	0.00	0.00	0.00
Vittadinia cuneata var. hirsuta	0.00	0.00	0.00	0.00	0.00	6.67	0.00	0.00	0.00	0.00
Vittadinia pterochaeta	0.00	16.67	0.00	17.65	0.04	40.00	0.01	66.67	0.00	25.00
Vittadinia sulcata	0.00	0.00	0.00	5.88	0.00	6.67	0.00	0.00	0.00	0.00
Wahlenbergia communis	0.00	8.33	0.00	47.06	0.17	60.00	0.00	16.67	0.00	0.00
Wahlenbergia gracilis	0.00	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Walwhalleya proluta	0.00	0.00	0.00	5.88	0.03	6.67	0.00	0.00	0.00	0.00
Xanthium occidentale	0.13	50.00	0.09	29.41	0.00	20.00	0.00	16.67	0.00	0.00
Xerochrysum bracteatum	0.00	58.33	0.00	29.41	0.03	20.00	0.00	0.00	0.00	0.00
Zaleya galericulata	0.02	16.67	0.03	23.53	0.17	13.33	0.17	33.33	0.00	0.00
Zygophyllum iodocarpum	0.00	0.00	0.02	11.76	0.00	0.00	0.00	0.00	0.02	75.00

Appendix 9: Average abundance and frequency of occurrence (%) of bird species in five vegetation types common to the lower Namoi floodplain

	River red gum		Coolibah		Grassland		Myall		Black box	
	Average abundance	Proportion of sites	Average abundance	Proportion of sites	Average abundance	Proportion of sites	Average abundance	Proportion of sites	Average abundance	Proportion of sites
Apostle bird	0.00	0.00	0.68	11.76	0.37	13.33	0.42	33.33	1.75	25.00
Australian hobby	0.04	8.33	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Australian king-parrot	0.08	8.33	0.00	0.00	0.00	0.00	0.33	16.67	0.00	0.00
Australian magpie	0.63	75.00	0.65	29.41	0.77	60.00	0.25	33.33	0.38	50.00
Australian pipet	0.00	0.00	0.03	5.88	0.03	6.67	0.00	0.00	0.00	0.00
Australian raven	0.67	58.33	0.53	35.29	0.17	26.67	0.17	33.33	0.25	25.00
Australian ringneck	0.08	8.33	0.59	35.29	0.17	26.67	0.17	16.67	0.25	25.00
Australian wood duck	0.17	16.67	0.06	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Bar-shouldered dove	0.17	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Black falcon	0.00	0.00	0.00	0.00	0.03	6.67	0.00	0.00	0.00	0.00
Black kite	0.29	33.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Black-faced cuckoo-shrike	0.25	16.67	0.00	0.00	0.23	26.67	0.00	0.00	0.25	25.00
Blue bonnet	0.00	0.00	0.09	5.88	0.60	26.67	1.33	50.00	0.50	50.00
Blue-faced honeyeater	0.13	16.67	0.00	0.00	0.07	6.67	0.00	0.00	0.00	0.00
Brown falcon	0.00	0.00	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Brown goshawk	0.00	0.00	0.03	5.88	0.00	0.00	0.08	16.67	0.00	0.00
Brown songlark	0.00	0.00	0.00	0.00	0.07	6.67	0.00	0.00	0.00	0.00
Brown treecreeper	0.21	16.67	0.35	29.41	0.00	0.00	0.00	0.00	0.00	0.00
Channel-billed cuckoo	0.00	0.00	0.00	0.00	0.03	6.67	0.00	0.00	0.00	0.00
Cockatiel	0.92	58.33	1.18	41.18	2.37	46.67	0.75	50.00	1.25	25.00
Common bronzewing	0.00	0.00	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Common starling	6.83	33.33	0.41	5.88	0.17	13.33	0.08	16.67	0.00	0.00
Crested pigeon	0.54	50.00	1.06	76.47	0.50	40.00	0.92	33.33	1.13	75.00
Dusky woodswallow	0.00	0.00	0.00	0.00	0.07	6.67	0.00	0.00	0.00	0.00
Eastern rosella	0.88	50.00	0.15	11.76	0.20	13.33	0.17	16.67	0.00	0.00
Eastern yellow robin	0.17	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Fairy martin	0.00	0.00	2.09	11.76	0.00	0.00	0.50	16.67	0.00	0.00

Galah	7.42	83.33	3.26	76.47	2.33	53.33	1.25	50.00	0.00	0.00
Glossy black-cockatoo	0.08	8.33	0.09	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Golden whistler	0.04	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Grey butcherbird	0.00	0.00	0.24	41.18	0.00	0.00	0.08	16.67	0.25	25.00
Grey fantail	0.38	33.33	0.26	23.53	0.00	0.00	0.42	16.67	0.25	25.00
Grey shrike-thrush	0.25	25.00	0.26	41.18	0.00	0.00	0.00	0.00	0.00	0.00
Grey-crowned babbler	0.46	8.33	0.00	0.00	0.10	6.67	0.00	0.00	0.00	0.00
Ground cuckoo-shrike	0.00	0.00	0.00	0.00	0.07	6.67	0.00	0.00	0.00	0.00
Horsfield's bronze-cuckoo	0.00	0.00	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Inland thornbill	0.04	8.33	0.00	0.00	0.00	0.00	0.08	16.67	0.00	0.00
Jacky winter	0.04	8.33	0.35	23.53	0.03	6.67	0.08	16.67	0.50	25.00
Kestrel	0.21	25.00	0.00	0.00	0.23	40.00	0.08	16.67	0.00	0.00
Kookaburra	0.17	33.33	0.21	29.41	0.10	13.33	0.08	16.67	0.00	0.00
Little corella	1.13	41.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Little cuckoo-shrike	0.21	16.67	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Little friarbird	1.38	41.67	1.06	47.06	0.00	0.00	0.42	50.00	0.38	25.00
Magpie-lark	0.54	50.00	0.38	35.29	0.43	20.00	0.50	33.33	0.38	25.00
Masked woodswallow	0.00	0.00	0.24	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Mistletoebird	0.00	0.00	0.15	17.65	0.00	0.00	0.50	50.00	0.00	0.00
Noisy friarbird	0.00	0.00	0.00	0.00	0.47	6.67	0.00	0.00	0.00	0.00
Noisy miner	1.83	75.00	2.65	47.06	0.60	20.00	0.00	0.00	0.00	0.00
Pacific black duck	0.00	0.00	0.06	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Pacific heron	0.00	0.00	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Pale-headed rosella	0.00	0.00	0.06	5.88	0.00	0.00	0.00	0.00	1.00	50.00
Pale-headed x eastern rosella	0.08	16.67	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Peaceful dove	0.58	33.33	0.50	29.41	0.00	0.00	0.00	0.00	0.00	0.00
Pied butcherbird	0.04	8.33	0.26	35.29	0.33	26.67	0.00	0.00	1.25	75.00
Rainbow bee-eater	0.00	0.00	0.06	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Red-capped robin	0.00	0.00	0.06	5.88	0.00	0.00	0.17	16.67	0.00	0.00
Red-rumped parrot	1.92	41.67	0.50	35.29	0.00	0.00	0.50	50.00	0.63	75.00
Red-winged parrot	0.29	25.00	0.00	0.00	0.00	0.00	0.17	16.67	0.00	0.00
Restless flycatcher	0.08	8.33	0.18	23.53	0.00	0.00	0.00	0.00	0.00	0.00
Rock dove	0.71	25.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Rufous songlark	0.33	16.67	0.21	17.65	0.00	0.00	0.08	16.67	0.00	0.00
Rufous whistler	0.08	8.33	0.15	11.76	0.00	0.00	0.00	0.00	0.25	25.00
Sacred kingfisher	0.13	25.00	0.21	17.65	0.00	0.00	0.00	0.00	0.00	0.00
Spiny-cheeked honeyeater	0.00	0.00	0.03	5.88	0.00	0.00	0.08	16.67	0.00	0.00
Spotted pardalote	0.08	16.67	0.03	5.88	0.00	0.00	0.00	0.00	0.00	0.00
Striated pardalote	0.04	8.33	0.09	17.65	0.27	13.33	0.08	16.67	0.00	0.00
Sulphur-crested cockatoo	7.58	100.00	0.24	17.65	0.33	26.67	0.58	50.00	0.00	0.00
Superb fairy-wren	0.38	33.33	0.35	17.65	0.13	6.67	0.58	33.33	0.00	0.00
Tree martin	0.92	16.67	0.38	5.88	0.00	0.00	0.25	33.33	0.00	0.00
Varigated fairy-wren	0.21	16.67	0.29	17.65	0.30	6.67	0.67	33.33	0.00	0.00
Weebill	0.25	16.67	1.00	41.18	0.00	0.00	0.33	16.67	1.25	25.00
Welcome swallow	0.08	8.33	0.15	11.76	0.13	20.00	0.08	16.67	0.00	0.00
Western warbler	0.04	8.33	0.06	5.88	0.00	0.00	0.00	0.00	0.38	25.00
Whistling kite	0.17	25.00	0.06	5.88	0.00	0.00	0.17	33.33	0.00	0.00
White-breasted sea eagle	0.00	0.00	0.15	11.76	0.00	0.00	0.00	0.00	0.00	0.00
White-breasted woodswallow	0.17	8.33	0.06	11.76	0.00	0.00	0.00	0.00	0.00	0.00
White-browed woodswallow	0.33	8.33	1.21	17.65	0.00	0.00	0.08	16.67	0.00	0.00
White-faced heron	0.17	16.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
White-plumed honeyeater	1.79	41.67	2.68	47.06	0.00	0.00	0.17	16.67	0.00	0.00
White-throated treecreeper	0.04	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
White-throated warbler	0.08	8.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
White-winged chough	1.08	41.67	0.56	11.76	0.00	0.00	0.25	16.67	0.00	0.00
White-winged fairy-wren	0.29	16.67	0.18	5.88	0.40	20.00	0.08	16.67	0.00	0.00
White-winged triller	0.00	0.00	0.18	23.53	0.00	0.00	0.00	0.00	0.00	0.00
Willie wagtail	0.46	33.33	0.41	35.29	0.00	0.00	0.33	16.67	0.75	25.00
Yellow thornbill	0.08	8.33	0.15	5.88	0.00	0.00	0.25	16.67	0.00	0.00
Yellow-rumped thornbill	0.00	0.00	0.03	5.88	0.00	0.00	0.17	16.67	0.00	0.00
Yellow-throated miner	0.08	8.33	0.29	17.65	0.27	20.00	1.67	66.67	4.63	100.00
Zebra finch	0.00	0.00	0.06	5.88	0.00	0.00	0.92	16.67	0.00	0.00