Fencing farm dams increases vegetation cover, water quality and macroinvertebrate biodiversity

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Abstract

In many farming landscapes, aquatic features such as wetlands, creeks and dams provide water needed for stock and irrigation, while also acting as habitat for a range of plants and animals. Indeed, some species threatened by land use change may otherwise be considerably rarer – or even extinct – in the absence of these habitats. Therefore, a critical issue for the maintenance of biodiversity in agricultural landscapes is the extent to which the management of aquatic systems can help promote the integration of agricultural production and biodiversity conservation. We completed a snapshot cross-sectional study in southern New South Wales (south-eastern Australia) to quantify the efficacy of simple management practices – partial revegetation and stock reduction via fencing – for improving vegetation structure, water quality, and macroinvertebrate assemblages. We found that even short-term livestock exclusion resulted in increased vegetation cover. Relative to dams that were unfenced, those that had been partially or completely fenced for many years were characterized by reduced turbidity and nutrient levels and contained fewer thermotolerant (faecal) coliforms. They also supported increased richness and abundance of macroinvertebrates. In contrast, control (unfenced) dams tended to have high abundance of a few macroinvertebrate taxa. Notably, differences remained between the macroinvertebrate assemblages of fenced dams and nearby ‘natural’ waterbodies. Our results show how management interventions can improve water quality in farm dams and provide a valuable reference and baseline for longer term studies of farm dam improvement.

Introduction

The majority of ecosystems worldwide are subject to some form of human intervention or management (IPBES, 2019). Although past land clearing and land use intensification have already caused extinctions (Maxwell et al., 2016), a critical step in preventing future biodiversity loss is to identify opportunities where conservation and agricultural production can co-occur (Leclère et al., 2020). While much research attention has focussed on the biodiversity values of uncleared vegetation within fragmented landscapes (Arroyo-Rodríguez et al., 2020; Haddad et al., 2017; Watson et al., 2018), there are cases where production activities themselves create novel habitats within otherwise intensively managed systems. For example, species that are adapted to early successional states can sometimes benefit from certain forms of timber harvesting (Swanson et al., 2011) or grazing regimes (Moranz et al., 2014). These habitats have the potential to support win-win outcomes where management can support both production and biodiversity.

Freshwater ecosystems are critical areas for biodiversity worldwide, but they are also highly threatened, with rivers having been regulated and wetlands drained and converted to other uses such as fields for agriculture (Reid et al., 2019). Where freshwater is retained in modified ecosystems, it often takes the form of artificial structures such as farm dams (Malerba et al., 2020). These artificial waterbodies can sometimes maintain substantial biodiversity, assuming that they are managed in a manner that maintains vegetation structure and water quality (Oertli, 2018). In farming systems, for example, artificial farm dams can support biodiversity in locations that would otherwise struggle to support diverse biotic communities (Chester and Robson, 2013; Hamilton et al., 2017; Hazell, 2003).

The Murray-Darling Basin, in south-eastern Australia, is the nation's most important food-producing area and supports more than 650 000 farm dams with more than 2.1 GL of water stored, primarily for domestic livestock (Srikanthan et al., 2015). Farms dams can be important for biodiversity conservation (Hamilton et al., 2017; Hazell et al., 2004, 2001), yet degraded dams can have significant negative impacts on the environment such as acting as a major source of greenhouse gas emissions (Ollivier et al., 2019). Restoration of farms dams to improve vegetation cover around and within them may not only reduce greenhouse gas emissions, but also improve water quality and, in turn, enhance the value of such areas for livestock production (Willms et al., 2002) and for biodiversity (Hamilton et al., 2017). However, there is currently limited information on biodiversity responses to management interventions to improve the condition of farm dams (Lewis-Phillips et al., 2019). Likewise, there is limited available data comparing the biodiversity of farm dams to that of local natural water bodies (Hazell et al., 2004).

To address these knowledge gaps, and to act as a baseline to longer-term studies of biodiversity responses in this ecosystem, we completed a cross-sectional field-based empirical study (sensu Cunningham and Lindenmayer, 2017) to compare the water quality and aquatic biodiversity of three categories of farm dams within a blocked design. These were: **(1)** 'business as usual' farm dams where there had been no attempts to improve environmental conditions, which we called 'control dams'; **(2)** farm dams where a range of environmental works were about to begin, or had just begun, termed 'transition dams'. And, **(3)** Dams that were partially or completely fenced to exclude stock, termed 'enhanced dams'. We also added a fourth category of connected ponds, often contained within creek lines, that represented the best available 'natural' state for our study region, and which provide a reference state against which to compare the effectiveness of restoration efforts.

We used the data gathered from the four kinds of water bodies on vegetation structure, water quality and macroinvertebrate assemblages to address two questions. First, we asked: *Does vegetation cover, water quality, and macroinvertebrate assemblage structure differ between* *fenced and unfenced dams, or natural waterbodies?* We anticipated that enhanced dams would have higher vegetation cover, lower turbidity and associated impurities, and higher invertebrate richness than control dams; but did not expect them to match natural controls in all respects. While our approach of comparing different waterbody types is useful, we also were interested to learn which aspects of restored dams most strongly influenced variation in macroinvertebrate abundance. This is important because it remains unclear to what extent macroinvertebrates respond directly to increased vegetation cover, versus a combination of increased vegetation cover and improved water quality. Therefore, we asked a second question: *What statistical associations link vegetation cover, water quality, and the abundance of macroinvertebrate taxa?* In combination, answering these two questions will provide new insights into the ecological properties of this regionally-significant landscape feature and how they might be modified in response to management interventions such as fencing (to limit access by domestic livestock). .

Methods

Study area and design

Our study region encompassed the agricultural landscapes of the South West Slopes Bioregion of NSW and North East Victoria (Fig. 1). This area is one of the most modified bioregions in Australia (Benson, 2008). The dominant land use is grazing of livestock for beef cattle, fat lamb and wool production, and dryland cropping of cereals and oilseed.

[Figure 1]

We surveyed 62 water bodies across 17 farming properties, in four categories. Enhanced dams (n = 21) were dams fenced to either exclude livestock entirely, or a provide a single hardened access point for the entry of livestock from an adjacent paddock. These sites have undergone revegetation of shrubs and trees. Sites selected under this category had been fenced for at least two years. Transition dams (n = 12) were those undergoing enhancement that have been recently fenced, totally or partially, for no longer than six months. Control dams (n = 24) were those that were unfenced and incorporated in the surrounding paddock. They were subject to the same management regimes as adjacent paddocks, either livestock grazing, dryland cropping or both. Natural waterbodies (n = 5) were naturally occurring and generally permanent water bodies. These were extremely rare in the landscape, hence our small sample size for this category. We targeted connected pond systems within creeklines for comparison to the static farm dam water bodies.

We established sites on commercial grazing (sheep/beef cattle) or mixed farming (dryland cropping and grazing) properties. We selected dams that were typically > 1 megalitre in capacity, as smaller dams were considered too ephemeral (i.e. they would likely dry up regularly). For each enhanced dam and each transition dam, we selected a control dam which had having similar characteristics, such as size, shape, position in landscape and surrounding land use. We selected matching treatment and control dams on the same farm, although some properties contained more than one waterbody in a given category. We selected a natural waterbody on or near each study farm wherever these occurred.

Field methods

We collected percentage cover data for vegetation attributes at three zones. Aquatic vegetation encompassed all vegetation within the waterbody itself, including submerged, floating and emergent vegetation. We classified vegetation as ‘riparian’ if it occurred between the high-water mark and actual water level at the time of survey, while ‘terrestrial’ vegetation included all vegetation from zero to twenty metres beyond the high-water mark.

We collected water samples at two metres from the edge of each water body and from a depth of 200mm. We avoided areas with floating debris and algae. For enhanced dams with livestock access points, we collected water samples adjacent to the access area. Samples were processed at Waterview Laboratory Howlong Road, Albury, NSW. Samples were tested for electrical conductivity, pH, chloride, total nitrogen (consisting of nitrate, nitrite, and Kjeldahl nitrogen), phosphorus, *Escherichia coli* and thermotolerant coliforms.

We sampled macroinvertebrates using a replicated edge sweep method (Gigney et al., 2007) at a subset of 29 waterbodies (14 enhanced dams, 11 control dams and four natural waterbodies). This involved four one metre (m) sweep searches representative of the habitats available, replicated three times across each site. We sorted samples in the field following the agreed level taxonomy (ALT) method, which allows for samples to be sorted in the field and avoids killing and preserving large numbers of specimens. In contrast to most existing methods that restrict identification to a specific taxonomic level (i.e. family), the ALT method classifies each taxon to the most precise taxonomic level that can be reliably identified in the field (Gooderham et al., 2010).

Statistical methods

Question 1: Does vegetation cover, water quality, and macroinvertebrate assemblage structure differ between fenced and unfenced dams, or *natural waterbodies?*

We used Generalised Linear Mixed Models (GLMMs) to quantify differences in vegetation cover, water quality, and macroinvertebrate assemblages between our four waterbody types. For vegetation cover, we calculated the proportion of area that was covered by any vegetation, then converted this to be unbounded by zero or one via the inverse logit transform, after setting values that were precisely zero or one to 0.001 or 1- 0.001, respectively. Conversely, we used the log transform for all of our water quality estimates (except pH), after removal of outliers (n = 4 values). We assumed a Gaussian error distribution for all of our vegetation cover and water quality models. Finally, we modelled species richness using a Poisson distribution and a log link. In all cases, we fit the same set of predictors; namely a fixed effect of waterbody type (a four-level factor) and a random effect of farm to account for the blocked design of our study. We conducted all of these analyses using the lme4 R package (Bates et al., 2014), and visualised the results using ggplot2 (Wickham, 2016), viridis (Garnier, 2018) and ggbeeswarm (Clarke and Sherrill-Mix, 2017) from the R statistical language (R Core Team, 2020).

Question 2: What statistical associations link vegetation cover, water quality, and the abundance of macroinvertebrate taxa?

To address our second question, we began by assuming a causal hierarchy between our different variables, and used this hierarchy to inform a set of models describing the potential associations between them. Specifically, we assumed that vegetation cover could be affected by waterbody type but not water quality or macroinvertebrates; water quality could be affected by waterbody type and/or vegetation cover; and that macroinvertebrates could be affected by any of the other three parameter sets. We then used GLMMs to build a set of competing models for each response variable, and used model selection (Burnham and Anderson, 2002) to choose a best model via the Bayesian Information Criterion (BIC). We maintained the transformations used in our previous stage of analysis, except that our invertebrate models had the abundance of a single invertebrate taxon as the response variable, the identity of the farm dam as a random effect, and a Poisson error structure with a log link.

The model sets were compared using BIC as follows. For vegetation cover, we simply used our earlier models of vegetation cover as a function of waterbody type and did not employ model selection. For water quality, we selected four response variables that best explained variation in the remaining set, as calculated using the ‘eleaps’ function in the R package ‘subselect’ (Orestes Cerdeira et al., 2020); these were pH, Chloride, total Nitrogen, and thermotolerant coliforms. We then compared: a null model with no fixed effects; a model that distinguished between dams and natural waterbodies; three models each containing a single vegetation cover variable (terrestrial, riparian or aquatic); and finally the additive or interactive effects of vegetation cover with waterbody type. Finally, for each invertebrate taxon found in >10 samples (n = 14), we ran 20 models using invertebrate abundance as our response variable. These models were specified as follows: a null model containing only an intercept and no predictors; seven models with only one term per model (waterbody type, vegetation structure in riparian or aquatic zones, or one of our four water quality measure); eight models with additive effects of water quality with vegetation structure; and four models with interactive effects of aquatic vegetation structure with water quality.

To present the results of this analysis, we began by using our first set of GLMMs to calculate predicted mean values of vegetation structure in each of our four waterbody types. We then used these predictions as inputs to our models of water quality variables; and then used those predictions as inputs to our GLMMs of invertebrate abundance. Finally, we compared each predicted value to the corresponding prediction for a control dam, enabling us to state how much a given parameter was higher or lower in the chosen waterbody type than we would expect in a control dam. This approach enabled us to plot a flow diagram of changes in key parameters for each waterbody type.

Results

Question 1: Does vegetation cover, water quality, and macroinvertebrate assemblage structure differ between fenced and unfenced dams*, or *natural waterbodies?

Vegetation cover surrounding our farm dams was typically highest in the terrestrial margin of the dam (mean = 86%), and declined in the riparian (54%) and aquatic (22%) zones. We also found a consistent pattern of lowest vegetation cover surrounding control dams, followed by transition dams, then enhanced dams, and finally the highest levels of cover were around natural waterbodies (Fig. 2). In combination, these results reflect a relatively small difference in terrestrial vegetation cover between control dams and natural waterbodies (85 – 96% as estimated using mixed models), but very large mean differences between control dams and natural waterbodies vegetation cover in the riparian (73% difference) and aquatic zones (96% difference).

[Figure 2]

Our water quality variables were highly correlated (Fig. S1), with four variables explaining 87% of the total information in the dataset: these were pH, chloride, total nitrogen, and thermotolerant coliforms. It was unsurprising, therefore, that groups of variables showed similar patterns of variation between waterbody types (Fig. 3). Specifically, variables associated with nutrient status (turbidity, nitrogen and phosphorus) and bacterial status (*E. coli*, thermotolerant coliforms) all had their highest values in control dams and their lowest in natural waterbodies, suggesting a positive influence of restored or natural waterbodies. Salinity variables (EC and chloride) did not vary greatly between farm dam categories but values were significantly higher in natural waterbodies. Finally, pH did not differ in a systematic way between waterbody categories, ranging from neutral to weakly alkaline in the majority of waterbodies.

[Figure 3]

In our macroinvertebrate analysis, we found that enhanced dams supported the largest numbers of macroinvertebrates, both in terms of species richness and total abundance (Fig. 4). Observed species richness within each sample ranged from three to 21 species, with the lowest mean richness in control dams (7.0 species) and the highest in natural waterbodies (13.8 species). These results were largely mirrored in our model of total abundance, with the exception that enhanced dams had the highest mean abundance (115.2 individuals), rather than natural waterbodies (97.2 individuals). In combination with our earlier findings, these results show that higher vegetation cover and reduced nutrients and bacterial pollutants in farm dams were associated with higher richness and abundance of macroinvertebrate taxa.

Question 2: What statistical associations link vegetation cover, water quality, and the abundance of macroinvertebrate taxa?

Model selection by BIC showed that both nitrogen and thermotolerant coliform levels were lower in waterbodies with higher cover of aquatic vegetation. Chloride levels were highest in natural waterbodies, and to a lesser extent, in sites with high terrestrial vegetation (though the latter effect was much weaker). Of the 23 macroinvertebrate taxa that were detected sufficiently often to enable statistical modelling, a model which included informative predictors (i.e. not the null model) was selected for 15 taxa (Fig. 5). pH was not shown to be affected by waterbody type or vegetation structure, and this variable was associated only with the abundance of one macroinvertebrate taxon (Diptera: Ceratopogonidae, or biting midges), and so for clarity we do not display this taxon in Figure 5.

We found that the most influential variable (in terms of number of species affected) was total nitrogen (n = 7 species) followed by percentage cover of riparian vegetation (n = 5 species). No other variable was associated with the abundance of more than two macroinvertebrate taxa, and terrestrial vegetation was not selected as a predictor for any taxa.

[Figure 5]

Combining predictions from the final models for all response variables (Fig. 5) showed that large increases in riparian vegetation associated with farm dams had a direct effect on macroinvertebrates, increasing occurrence of four taxa and reduced Annelid abundance in the dam. The increase in aquatic vegetation in enhanced dams had a greater influence on macroinvertebrates than the increase in riparian vegetation, despite being of lower magnitude. This was because aquatic vegetation was also associated with reduced nutrient levels that was strongly correlated with abundance of a number of macroinvertebrate taxa. Specifically, enhanced dams supported 45% less nitrogen than control dams on average, and this decrease was itself associated with an increase in abundance of Orders Odonata, Trombidiformes and Decapoda, as well as a 93% decline in Static Boatmen (Genus Agraptocorixa) that were a dominant part of the assemblage in control dams. Increased aquatic vegetation was also associated with decreased coliform levels, which is desirable in itself; but also had a positive effect on the abundance of Chiromonidae (non-biting midges). Increased aquatic vegetation was associated with a small (10%) increase in leech abundance (subclass Hirudinea).

Discussion

A growing body of research has demonstrated the value of farm dams for biodiversity conservation in agricultural landscapes (Brainwood and Burgin, 2009; Hamilton et al., 2017; Hazell et al., 2001; Lewis-Phillips et al., 2020; Reyne et al., 2020). However, there has been only limited work to date that quantifies the outcomes of management interventions aimed at enhancing the condition and ecological values of Australian farm dams (Hazell et al., 2004; Lewis-Phillips et al., 2019). Similarly, there has been relatively limited work on how the ecological values of enhanced dams compared to natural water bodies present in the same landscape (but see Reyne et al., 2020). We sought to address these knowledge gaps in a comparative study conducted in south-eastern Australia. Our empirical study led to three key findings. These were: **(1)** Fencing of farm dams to limit livestock access resulted in major changes in aquatic vegetation as well as a range of variables associated with water quality. **(2)** Levels of *E. coli* and thermotolerant (faecal) coliforms were extreme in some dams, exceeding safe levels (as determined by ANZECC and ARMCANZ, 2000) by over an order of magnitude. And, **(3)** Most macroinvertebrate taxa were more abundant in enhanced dams relative to control dams, but changes in abundance were not related to their indicator value in flowing waters (as determined using the ALT measure), indicating that a modified indicator schema may be needed for farm dams (see also Chessman et al., 2002). In the remainder of this paper, we further discuss these key findings and their significance for farm and wetland management in our study region.

Response to management interventions

Our key finding was that fencing to reduce or exclude livestock from farm dams, combined with revegetation of terrestrial plants at some dams, was associated with marked improvements in both vegetation cover and water quality. In addition, we found a strong association between interventions to enhance farm dam condition and high levels of taxonomic richness of macroinvertebrates and a number of individual taxonomic groups. Combining models of vegetation, water quality, and macroinvertebrate taxa revealed greater macroinvertebrate abundance in enhanced dams (Fig. 4) was associated with a combination of increased vegetation cover and reduced turbidity and nutrient levels (Fig. 5). While our study did not investigate the influence of these changes on taxa such as frogs, reptiles or birds, there are examples where increases in the abundance of aquatic invertebrates have been shown to support populations of vertebrate predators (Lewis-Phillips et al., 2020). Further, improvements in aquatic vegetation similar to those documented here have been shown to have a direct positive effect on a broad range of species and taxonomic groups such as zooplankton (Le Quesne et al., 2020) and frogs (Hazell et al., 2001). Overall, therefore, our results support restricting livestock access to dams through complete or partial fencing as an effective method of improving the environmental performance of farm dams.

Our macroinvertebrate surveys showed that although farm dam enhancement was associated with an overall increase in the abundance and taxonomic richness of macroinvertebrate assemblages, in practice each taxon responded to different aspects of dam rehabilitation. We found an association between increased levels of riparian vegetation and lower levels of

Annelids, presumably because for the filtering effect of riparian vegetation during rainfall. In addition, increased aquatic vegetation was indirectly associated with an increase in yabbie numbers via a link to reduced nitrogen (Fig. 5).

Water quality in farm dams

Our most concerning finding was that levels of faecal coliforms and *E. coli* were extremely high in some dams, and particularly in unfenced control dams. In Australia, standard guidelines for livestock drinking water quality recommend that thermotolerant coliform counts do not exceed 100 organisms/100mL for livestock drinking water (ANZECC and ARMCANZ, 2000). While this guideline is intended to be indicative rather than rigidly enforced, it is nonetheless informative that this threshold was exceeded in approximately 65% of control dams in our study. Even more concerning is that the peak value recorded in our study was over two orders of magnitude higher than this threshold (n = 24,196), while 16 dams had thermotolerant coliform values at least an order of magnitude higher than the threshold (i.e. 1000 coliforms/100mL). What is less clear, however, is what the implications of such high levels might be for stock health. Microbial pathogens have been shown to have negative effects on animal performance (Anderson, 1987); but there is evidence that cattle can tolerate high levels of microbial flora (Lardner et al., 2005; Willms et al., 2002). More concerning is that faecal contamination can affect the palatability of water, and therefore water consumption by cattle (Holechek, 1979; Willms et al., 2002), potentially leading to dehydration and inefficient rumination leading to reduced stock condition and productivity. More encouraging was our finding that levels of thermotolerant coliforms and *E. coli* were both lower on average in enhanced dams than in control dams, although there was only a large difference for *E. coli* counts (Fig. 3g). Interestingly, transition dams also exhibited a significant reduction of thermotolerant coliforms relative to control dams (Fig. 3h), despite having excluded stock for a very short period of time (< 6 months), suggesting that reductions in pathogens happens rapidly once stock are excluded.

Beyond our concerning findings regarding water-borne bacteria, it remains challenging to classify what the environmental and production impacts might be as a consequence of poor water quality in unfenced dams. The majority of water quality indices that we measured either did not have an accepted standard safety limit for stock that we could find (e.g. phosphorus), or values did not exceed those limits (e.g. salinity). One point not investigated by our analysis, but that wold be worthy of further study, is the risk of biotic effects such as growth of toxic algae that can both reduce palatability of water for livestock (Hyder and Bement, 1968) and potentially impact animal health (Steffensen et al., 1999). An associated environmental risk is that eutrophic dams can release large quantities of greenhouse gases. Indeed, in a recent study, (Ollivier et al., 2019) showed that farm dams adjacent to our study region contribute an order of magnitude more methane than comparable freshwater lakes and reservoirs. However, they also showed that CO_2-equivalent emissions were dramatically reduced in dams with lower nitrate levels, which is encouraging given our finding that fenced dams have lower nutrient levels (Fig. 3) than unfenced dams, likely due to their higher coverage of aquatic and riparian vegetation (Fig. 2).

Natural vs anthropogenic waterbodies

Finally, our work revealed that although enhanced dams displayed many similar properties to natural water bodies, suggesting that management interventions can promote a successful transformation to a better functioning freshwater ecosystem, there also were some large differences. For example, chloride levels and percentage cover of aquatic vegetation, which were both higher in natural water bodies than found in enhanced dams. Higher chloride may be a direct result of natural water bodies occurring in lower parts of the landscape, where chloride may accumulate. The higher average percentage cover of aquatic vegetation in natural water bodies may be influenced by differences in geometry relative to enhanced dams (and farm dams in general), with the latter having a much larger surface area and containing areas of much deeper water. Notably, other studies have found inherent differences in several key attributes between natural water bodies and artificial water bodies such as farm dams (Hazell et al., 2004; Le Quesne et al., 2020; Reyne et al., 2020). Such differences suggest that natural water bodies may not be an entirely appropriate benchmark targets for guiding the restoration of farm dams to improve their condition, water quality and ecological value for biodiversity.

Conclusions

Overall, our results documented a significant improvement in water quality and biodiversity resulting from farm dam enhancement, with some effects becoming evident within a relatively short period of stock exclusion and revegetation (<6 months). This result gives confidence that these improved outcomes will flow on to a broader suit of taxa, not measured in this study. Further work to quantify the effect of farm dam enhancement on other taxonomic groups is required, as is longer-term research to understand the role of variation in climate on farm dams. In addition, it is possible that water quality improvements resulting from enhancing farm dams could improve domestic livestock health and productivity (Willms et al. 2002), although experimental local trials would be necessary to substantiate these claims.

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Figure captions

Figure 1: Map of the study region, with study farms shown as points.

Figure 2: Proportional vegetation cover in three zones (terrestrial, riparian and aquatic) between our four waterbody types.

Figure 3: Water quality measures by waterbody type, showing mean and 95% confidence intervals from Linear Mixed Models (LMMs). Note all plots are shown on a log(y) scale, but the model for pH was calculated without a log transformation.

Figure 4: Macroinvertebrate richness (a) and abundance (b) across the four waterbody types.

Figure 5: Expected values of vegetation structure, water quality and macroinvertebrate abundance in enhanced dams (a) and natural waterbodies (b). Lines show positive (red) or negative (blue) effects of variables selected by BIC, while numbers in parentheses show the difference in the expected value of that parameter from the expected value for a control dam. One invertebrate group (Diptera: Ceratopogonidae) has been removed from the diagram for clarity (see text).

Data Accessibility

Raw data for all analyses in this paper will be made available via Dryad.

Competing Interests

The authors declare no competing interests.

Author Contributions

All authors contributed to study design; MC, CC, CO, AS & DS collected the data; MW, MC, BS & DBL wrote the first draft; all authors contributed to revisions and approved the submission.

Appendices

Figure S1: Correlation matrix for water quality variables

[Figure S1]